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High Rate Nutrient Removal for Combined Sewer Overflows. Bench Scale and Demonstration Scale Studies

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High Rate Nutrient Removal for Combined Sewer Overflows

Bench Scale and Demonstration Scale Studies

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The results of this program have demonstrated that the simultaneous removal of suspended solids, phosphorus, and ammonia-nitrogen from a combined sewer overflow is feasible using the high rate unit process concept.

Suspended solids removal ranged from 90-100% with alum and polymer dosages of 220 mg/l and 1 mg/l, respectively.

Alum dosages resulting in Al/P molar ratios greater than 1.4 were effective in reducing the total inorganic phosphorus (TIP) concentrations from greater than 10 mg/l to less than 0.9 mg/l.

The clinoptilolite was effective in reducing the ammonia level to below the detectable levels of 0.02 mg/l NH_3-N. Infulent NH_3-N concentrations ranging from 0.2 to 16 mg/l were reduced below detectable levels during the initial period of contact. The effective exchange capacity of clinoptilolite was found to range from 0.20 to 0.64 meq NH_3-N/g clinoptilolite as the influent NH_3-N concentration ranged from 7.5 to 16 mg/l, respectively.

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HIGH RATE NUTRIENT REMOVAL FOR COMBINED SEWER OVERFLOWS

Bench Scale and Demonstration Scale Studies

by

C. B. Murphy, Jr., Orest Hrycyk, William T. Gleason
O'Brien & Gere Engineers, Inc.
Syracuse, New York 13221

Grant No. S-802400

Project Officer

Chi-Yuan Fan
Storm and Combined Sewer Section
Wastewater Research Division
Municipal Environmental Research Laboratory (Cincinnati)
Edison, New Jersey 08817

MUNICIPAL ENVIRONMENTAL RESEARCH LABORATORY
OFFICE OF RESEARCH AND DEVELOPMENT
U.S. ENVIRONMENTAL PROTECTION AGENCY
CINCINNATI, OHIO 45268

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FOREWORD

The U.S. Environmental Protection Agency was created because of increasing public and governmental concern about the dangers of pollution to the health and welfare of the American people. Noxious air, foul water, and spoiled land are tragic testimony to the deterioration of our natural environment. The complexity of that environment and the interplay between its components require a concentrated and integrated attack on the problem.

Research and development is that necessary first step in problem solution and it involves defining the problem, measuring its impact, and searching for solutions. The Municipal Environmental Research Laboratory develops new and improved technology and systems for the prevention, treatment, and management of wastewater and solid and hazardous waste pollutant discharges from municipal and community sources, for the preservation and treatment of public drinking water supplied and to minimize the adverse economic, social, health, and aesthetic effects of pollution. This publication is one of the products of that research; a most vital communications link between the researcher and the user community.

One of the many areas of concern in the water pollution field is the effect of combined sewer overflows on the water quality of the receiving streams. This report is the culmination of a study which piloted a unique concept in treating combined sewer overflows for the removal of the macronutrients, phosphorus and ammonia nitrogen. The high rate physical/chemical system consisted of in-line chemical addition, mixing, coagulation and flocculation, filtration and ion exchange. The treatment system described herein, lends itself to situations where land availability is at a minimum and storage or conveyance of combined sewer overflows to existing treatment sites is impractical.

Francis T. Mayo
Director
Municipal Environmental Research
Laboratory

ABSTRACT

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This study was jointly supported by the United States Environmental Protection Agency, Grant Number S-802400, and the Department of Drainage and Sanitation of Onondaga County, New York.

Bench scale work was commenced in January 1974. Construction of demonstration scale facilities began shortly after the completion of the bench scale work in 1975. The demonstration scale facility was started up in June 1975 and terminated in August 1976.

CONTENTS

Foreword....iii
Abstract.... iv
Figures.... vi
Tables....vii
Abbreviations and Symbols....viii
Acknowledgement.... ix

1. Introduction....1
2. Conclusions....3
3. Recommendations....7
4. Bench Scale Study - Results and Discussion
   - Media Selection....8
   - Coagulation & Flocculation Studies....9
   - Clinoptilolite Evaluation....10
   - Phosphorus Removal....14
   - Ammonia Removal....15
5. Demonstration Pilot Plant-Experimental Design & System Description
   - Design Philosophy....21
   - Equipment Description....25
6. Demonstration Pilot Plant-Results and Discussion
   - Combined Sewer Overflow Application....30
   - Dry Weather Creek Flow Treatment....47
7. Regeneration Evaluation
   - Clinoptilolite Regeneration....49
   - Alum Regeneration....53
8. Projected System Costs....55

References....60

Glossary....62

FIGURES

Number		Page
1	Ammonia Removal Efficiency	16
2	Effect of Clinoptilolite Depth on Ammonia Removal Efficiency	17
3	Effect of Clinoptilolite Particle Size on Ammonia Removal Efficiency	19
4	Effect of Clinoptilolite Particle Size and Depth on Ammonia Removal Efficiency	20
5	Maltbie Street Site Plan	26
6	Process Flow Diagram	27
7	Microwedge Strainer Assembly for Filters	29
8	Suspended Solids Removal Run #2	30
9	Suspended Solids Removal Run #5	32
10	Suspended Solids Removal Run #22	32
11	Suspended Solids Removal Run #28	33
12	Suspended Solids Removal Run #36	33
13	Percent Suspended Solids Removal vs Influent Suspended Solids	34
14	Percent Suspended Solids Removal vs Influent Suspended Solids	35
15	Effluent vs Application Rate	36
16	Percentage of Phosphorus Remaining	37
17	Phosphorus Removal	37
18	Phosphorus Removal Run #9	38
19	Percentage of Phosphorus Remaining	38
20	Residual Phosphorus Concentration	39
21	Ammonia-Nitrogen Removal Breakthrough Curve	40
22	Ammonia-Nitrogen Removal Breakthrough Curve First Testing Series	41
23	Ammonia-Nitrogen Removal Breakthrough Curve Second Testing Series	42
24	Ammonia-Nitrogen Removal Breakthrough Curve Third and Fourth Testing Series	45
25	Zeolite Regeneration System	49
26	Ammonia Recovery from Thermal Regeneration of Clino	51
27	X-ray Diffraction Patterns of Clinoptilolite	52

TABLES

Number		Page
1	Effective Exchange Capacity of Clinoptilolites From Various Deposits	4
2	Polyelectrolyte Evaluation	9
3	Effective Exchange Capacities of Clinoptilolite Samples	12
4	Effective Exchange Capacities of Clinoptilolite Samples	12
5	Effect of Pretreatment on the EEC of Clinoptilolite	14
6	Effect of Clinoptilolite Depth on Ammonia Removal Efficiency	16
7	Effective Exchange Capacity of Clinoptilolite Under Constant Flow Conditions	18
8	Clinoptilolite Particle Size Distribution After Backwashing	24
9	Velocity Gradients for High and Low Energy Mixers	28
10	Selected Data for Combined Sewer Overflow Runs	31
11	Additional Analyses for A Combined Sewer Overflow	46
12	Effective Adsorption Capacity of Thermally Regenerated Clino	51
13	Operating Characteristics For a 10 MGD Combined Sewer Overflow Nutrient Removal Facility	56
14	Capital Costs For 10 MGD Nutrient Removal System for CSO	56
15	Annual Cost of Nutrient Removal Facility	58

LIST OF ABBREVIATIONS AND SYMBOLS

ABBREVIATIONS

BV/hr	Bed volume per hour
cfm	Cubic feet per minute
clino	Clinoptilolite
CSO	Combined Sewer Overflow
F Coli	Fecal coliform bacteria
fps/ft_2	Feet per second per foot
gpm/ft^2	Gallons per minuted per square foot
Kg/cm^2	Kilogram per square centimeter
KWH	Kilowatt hour
$l/min/m^2$	Liter per minute per square meter
m^3/min	Cubic meters per minute
meq/g	Milliequivalents per gram
mgd	Million gallons per day
mg/l	Milligrams per liter
ml	Milliliter
mm	Millimeter
N	Normality
psi	Pounds per square inch
SS	Suspended solids
TIP	Total inorganic phosphorus
TOC	Total organic carbon
EEC	Effective ammonia exchange capacity

SYMBOLS

Al	Aluminum
$Al(OH)_3$	Aluminum hydroxide
$AlPO_4$	Aluminum phosphate
$Al_2(SO_4)_3 . 14 H_2O$	Alum
Ba	Barium
Ca	Calcium
CO_2	Carbon dioxide
Cr	Chromium
Cu	Copper
Fe	Iron
HCl	Hydrochloric Acid
HCO_3	Bicarbonate
H_2SO_4	Sulfuric Acid
K	Potassium
Mg	Magnesium
Na	Sodium
NaCl	Sodium Chloride
NaOH	Sodium Hydroxide
Na_3PO_4	Sodium phosphate
NH_3-N	Ammonia as nitrogen
P	Phosphorus
PO_4^{-3}	Phosphate
P_2O_5	Phosphorus pentoxide
Zn	Zinc

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The project was administrated under the direction of John M. Karanik, Grant Director, County of Onondaga, and Frank J. Drehwing, Vice President, O'Brien and Gere Engineers Incorporated. Technical supervision was provided by C. B. Murphy, Jr., Managing Engineer, and Orest Hrycyk, Project Manager.

The bench-scale studies were performed in the pilot testing area of the offices of O'Brien and Gere Engineers, and the assistance of all personnel who performed countless chemical analyses is acknowledged.

SECTION 1

INTRODUCTION

Most older cities in the United States have a combined sewer system that serves dual purposes: (1) continuous conveyance of domestic sewage from households, commercial and industrial institutions, and (2) conveyance of intermittent storm runoff during times of precipitation, snow melt, and street washings. During times of heavy precipitation, the increased flow in the sewers can result in severe hydraulic overloading of the wastewater treatment plant. The result of this overloading can be reduced treatment efficiency and the possibility of a biological system washout.

To prevent such occurrences, combined sewer systems were designed with system relief points which divert the excess flow from the conveyance and treatment system to a holding basin temporarily or the nearest body of surface water. This places a severe wet weather pollutional load on the receiving streams. Since precipitation cleanses homes, cars, streets, commercial, industrial, and agricultural areas, stormwater runoff contains substantial quantities of nutrients, organics, salts, solids, heavy metals, oils, pesticides, herbicides, etc. Several studies which characterized the chemical and physical nature of stormwater runoff have singled out untreated stormwater overflows as a significant threat to surface water quality. (1) (2) (3)

Organic material, both soluble and insoluble, exerts an oxygen demand in the receiving stream which can create oxygen limiting conditions in the water column, thereby impairing the level of aquatic life. Excess quantities of organics, and more importantly, excess inorganic nutrients can accelerate the onset of eutrophication, seriously disrupting the character of the aquatic system. Increases in suspended matter, heavy metals, and in the bacterial population all exert a detrimental effect on the receiving stream.

This study is primarily concerned with the evaluation of a single unit process for the removal of suspended solids and the macronutrients (phosphorus and nitrogen) from the combined sewer overflows.

Uttormark, et al. (4) estimate that runoff from urban areas has a higher nutrient concentration than runoff from other non-point sources such as agricultural areas, forests, and wetlands. Murphy (5) in a study of the nutrient levels in Onondaga Lake, Syracuse, N.Y., and its tributaries estimates that of the total inorganic phosphorus entering the lake, 71 percent is from municipal continuous discharges and 19 percent is from municipal intermittent (wet weather) sources such as combined sewer overflows.

Phosphorus has long been suspected as a limiting nutrient for sustaining algal growth. In lake waters slight increases in the phosphorus concentration are known to overstimulate the growth of algae resulting in algal blooms. However, these nuisance blooms can be controlled and in some cases, eliminated. Murphy (5) noted that a reduction of the total inorganic phosphorus concentration in Onondaga Lake from 1.74 mg/l to 0.74 mg/l resulted in a decline and eventual elimination of the bluegreen APHANIZOMENON, which in previous years was dominant in the late summer and early fall growing seasons. It was also noted that as a result of reducing the phosphorus concentration, the phytoplankton diversity index increased, reflecting a higher degree of stabilization within the phytoplankton community.

Nitrogen, like phosphorus,is one of the macronutrients required for algal growth. However, only a few species of algae can satisfy their requirement by direct use of molecular nitrogen. Typically, most algae use ammonia, nitrite or nitrate,as a nitrogen source for their synthesis of proteins and other forms of organic nitrogen. As in the case with phosphorus, an overabundance of nitrogen (ammonia, nitrite, or nitrate) in a surface water can trigger an algal bloom.

The objective of the project was to determine the effectiveness and feasibility of dual use of a high rate physical chemical process for removal of the suspended solids, phosphorus, and ammonia nitrogen from combined sewer overflows and creek flow during wet-weather and dry-weather, respectively.

The project consisted of two phases: (1) a bench scale study, and (2) a demonstration scale study. Both studies used in-line coagulation and flocculation, high rate filtration and ion exchange. Due to several operational restrictions at the demonstration site, the larger scale study used the two media in separate contactors rather than in a single contactor. The soluble and particulate phosphates were precipitated and entrained in an alum floc and removed along with other suspended and coagulated matter in the primary filter medium, while the soluble ammonia and short chain organic nitrogen species were removed by ion exchange mechanisms in the secondary filter medium, a natural occurring zeolite, clinoptilolite.

The demonstration scale study was designed to verify the results of the bench study and to develop treatment unit costs.

SECTION 2

CONCLUSIONS

1. The results of this program have demonstrated that the sequential removal of ammonia and phosphorus from a combined sewer overflow is feasible using a high rate unit process. The phosphorus is removed via alum addition, coagulation, and filtration. Ammonia removal is accomplished via an exchange/adsorption mechanism using a natural occurring zeolite, clinoptilolite, which also serves as a polishing filter.

2. The in-line addition of alum and polymer and subsequent in-line coagulation, flocculation and high rate filtration, at a surface application rate of 407 l/min/m^2 (10 gpm/ft^2), through a dual media process consisting of granular coal material and a naturally occurring zeolite, produced a high quality effluent. An alum dosage of 110 mg/l when combined with a polymer dosage of 1.0 mg/l was capable of reducing the influent suspended solids level from 50 to 400 mg/l to below 40 mg/l.

3. The added alum was effective in reducing phosphorus levels in the applied wastewater. An aluminum to phosphorus molar ratio of 1.4 or greater resulted in effluent phosphorus concentrations below 0.5 mg/l.

4. High rate filtration of the combined sewer overflow with alum addition resulted in several additional benefits. Test results indicated that in addition to the reductions in phosphorus and suspended matter, reductions in TOC, heavy metals and fecal coliform could also be expected. TOC reductions averaged 50%, whereas, heavy metal reductions averaged 99+% for chromium and iron, and 50% for zinc and copper. In a number of cases, a 2.5 log reduction in fecal coliforms was also realized.

5. Effective backwashing was achieved by preliminary air scouring at 0.11 m^3/min (4 cfm) for 8 minutes. Backwashing at 814 l/min/m^2 (20 gpm/ft^2) was commenced during the final three minutes of air scouring and continued for 15 minutes. With the use of air scour, 4 to 8 percent of the filtered volume was required for backwashing.

6. Clinoptilolite, a naturally occurring zeolite, is capable of reducing the ammonia levels in combined sewer overflows to less than 0.02 mg/l ammonia nitrogen (NH_3-N). Utilizing fresh clinoptilolite influent NH_3-N concentrations as low as 0.2 mg/l can be reduced to below the detectable limits during the initial contact period.

Clinoptilolite's ammonia removal capacity was determined to be a function of the NH_3-N concentration in the contacting solution. The effective exchange capacity of clinoptilolite was found to range from 0.20 to 0.64 meq NH_3-N/g of clinoptilolite as the influent NH_3-N concentration ranged from 7.5 mg/l to 16 mg/l NH_3-N, respectively.

7. Improvement in the effective exchange capacity (EEC) of clinoptilolite was observed when decreasing the flow rates from 14 BV/hr to 2.8 BV/hr. The EEC's were determined to be 0.64 and 0.42 meq NH_3-N/g clinoptilolite for the 2.8 and 14 BV/hr flow rates, respectively. Although the flow rate decreased by a factor of 5, there is only a 48% increase in EEC. This percentage increase in the EEC does not justify designing a system at the lower flow rate.

8. The clinoptilolite has the ability to reduce any high peaks in the influent NH_3-N concentration even after the effective exchange capacity has been exceeded. However, ammonia leaching from the clinoptilolite occurs when the influent NH_3-N concentration drops below the equilibrium concentration.

9. Considerable variability in the effective exchange capacity is exhibited by various types of clinoptilolite. Evaluations of samples from eight deposits in western United States revealed a wide variation in the EEC. EEC values are presented in Table 1. Hector clinoptilolite (San Bernardino County, CA), was used exclusively in this project, since it was the only clinoptilolite that was commercially available in the quantities required.

TABLE 1: EFFECTIVE EXCHANGE CAPACITY OF CLINOPTILOLITES FROM VARIOUS DEPOSITS

Origin of Clinoptilolite Sample	Effective Exchange Capacity meq NH_3-N/g Clinoptilolite
1. Malheur County, Oregon	0.390
2. Owyhee County, Idaho	0.343
3. Grant County, New Mexico	0.331
4. San Bernardino County, California (Hector)	0.311
5. Lander County, Nevada	0.250
6. Fremont County, Wyoming	0.187
7. Maricopa County, Arizona	0.162
8. Washoe County, Nevada	0.146

NH_3-N conc. of contacting solution = 21 mg/l

10. The effect of chemical pretreatment was evaluated for five of the eight clinoptilolite samples listed in Table 1. Clinoptilolite samples 1 and 7 in Table 1 exhibited an increase in the EEC after treatment with 6N HCl followed by a caustic brine rinse, whereas, samples 2, 4 and 8 exhibited a decrease in the EEC.

11. Phosphorus removal results indicated that in-line static mixers are effective in providing the necessary high energy mixing required for the initial aluminum and phosphorus contact.

12. Al:P molar ratios above 1.4 were necessary to consistently obtain effluent TIP concentrations within the range of 0.2 to 0.9 mg/l. Effluent TIP concentrations increased dramatically as the Al:P molar ratio decreased below 1.4.

13. Treatment of creek water resulted in ammonia and phosphorus removal efficiencies similar to those obtained when treating a combined sewer overflow.

14. Thermal regeneration of clinoptlolite is a feasible alternative to chemical regeneration. The optimum regeneration temperature is 500°C to 540°C with a regeneration time of one hour. Lower temperatures do not result in complete clinoptilolite regeneration and temperatures greater than 550°C appear to cause some structural alterations which affect the ammonia exchange capacity.

15. Thermal regeneration of clinoptilolite is an energy intensive operation, and as such, is more expensive than conventional regeneration. It is estimated that the energy cost for thermal regeneration of clinoptilolite (with 40% moisture) adds an additional $\$0.023/m^3$ ($0.088/1000 gal) to treatment costs for a wastewater containing 10 mg/l NH_3-N. According to Koon and Kaufman (6), the least cost of chemical regeneration was achieved when the regenerant was reused. This cost was estimated to be $\$0.013/m^3$ ($0.050/100 gal) for a wastewater containing 20 mg/l NH_3-N. Reducing the influent NH_3(N) concentration to 10 mg/l would increase the resultant cost.

    Thermal regeneration can be cost-effective if a source of waste heat, such as that liberated from a carbon regeneration facility, is in reasonable proximity to a treatment system using clinoptilolite.

16. Clinoptilolite which was subjected to 3 cycles of regeneration at 500°C and subsequent ammonia uptake still retained greater than 97% of its original EEC.

17. Essentially, 90% of the ammonia adsorbed onto the clinoptilolite is removed after one hour of thermal regeneration at 500°C. This was verified by using argon gas to transport the desorbed ammonia to a trap.

18. Thermal treatment of waste alum sludge at 1000°C had no significant effect on the amount of phosphorus that was reclaimed with the alum. This indicates that little P_2O_5 was lost upon thermal regeneration. Thermally reclaimed aluminum sulfate from waste alum sludge appears to be unsuitable for use as a coagulant where phosphorus removal is required.

19. The high rate nutrient removal system was also found to be successful in reducing nutrient concentrations from the dry-weather flow of a tributary

Onondaga Creek, discharging to Onondaga Lake. Multiple application would make better use of the wet-weather facilities and would provide nutrient reductions preceding critical periods of phytoplankton production.

20. The capital costs for a nutrient removal facility designed to treat 10 mgd of CSO are estimated to be $4.9 million (ENR Construction Cost Index 2573).

21. Assuming 100 overflow occurrences are treated per year, the annual operating, maintenance, and amortization costs of a 10 mgd nutrient removal facility with thermal regeneration of spent clinoptilolite are expected to be $0.7 million.

SECTION 3

RECOMMENDATIONS

1. The ammonia removal application of a wide range of natural zeolites including clinoptilolite should be evaluated for the purpose of establishing a source of material having a high ammonia exchange capacity and optimal resistance to mechanical degradation.

2. A long term study utilizing the configured nutrient removal process should be conducted to determine variability in ammonia removal efficiency, mechanical attrition and the effects of oil and greases and divalent metal cation species under system optimized operating conditions.

3. The operation of the nutrient removal process should be evaluated under a dual dry-and wet-weather operation. The dry-weather application should involve a tertiary treatment application on conventional sanitary wastewater.

4. The biological regeneration of spent clinoptilolite should be evaluated. Particular attention should be given to the definition of optimized aeration requirements, contact time, bioregeneration of kinetics, and physical attrition of the exchange capacity.

5. The thermal regeneration of spent clinoptilolite should be investigated on a scaled up basis with particular emphasis on establishing design parameters.

SECTION 4

BENCH SCALE STUDY RESULTS AND DISCUSSION

MEDIA SELECTION

Prior to any treatment evaluations it was necessary to determine the optimum particle size of Anthrafilt (trade name for anthracite filter media) to be used with a -20+50 mesh clinoptilolite. This particle size of clinoptilolite was selected because it offered the highest surface area and exchange capacity, at an acceptable headloss when used in a column. A -20+50 mesh material is one which passes through a No. 20 U.S. Standard Sieve, aperture opening 0.84 mm (0.033 in.), but is retained on a No. 50 Sieve, aperture opening 0.30 mm (0.012 in.).

Compatability evaluations were conducted by backwashing a 15.25 cm (6 in.) by 3.65 m (12 ft.) plexiglas column filled with both media, and observing the particle distribution pattern after settling. That anthrafilt which produced a minimum interphase with the clinoptilolite would be used in the bench scale testing. Anthrafilt sizes investigated were numbers 1-1/2, 2 and 3 whose particle sizes were 0.85 to 95 mm, 2.38 to 4.76 mm and 4.76 to 7.94 mm, respectively.

These tests revealed the incompatability of the -20+50 mesh clinoptilolite with any of the aforementioned Anthrafilt sizes. Upon backwashing, the lighter clinoptilolite particles sifted upward through the larger Anthrafilt particles which were settling, resulting in an inversion of the media. Since it was not possible to find an Anthrafilt size that was compatable with the -20+50 mesh clinoptilolite, the clinoptilolite particle sizes were increased and tested with the aforementioned Anthrafilt sizes. Additional testing revealed that a -4+20 (<4.76 mm >0.84 mm) mesh clinoptilolite and No. 1-1/2 Anthrafilt resulted in no media inversion and produced only a 2.5 cm (1 in.) zone of media integration after backwashing.

Although these two media were found to be compatible, it was felt that using them in an actual filter operation would not be desirable. The disadvantages would include the fact that the small particle sizes found in the No. 1-1/2 Anthrafilt increase the probability of the floc blinding the media in a shorter timeframe thereby resulting in higher head losses and shorter filter runs. Secondly, the use of the -4+20 mesh clinoptilolite would result in a dramatic decrease in the ammonia removal efficiency, since according to other investigations (7) it was determined that the ammonia removal capacity of clinoptilolite is inversely proportional to the particle size.

It was for the above reasons that a search for a filter medium, with a density less than Anthrafilt was initiated. Several synthetics (plastics) were evaluated as to their specific gravity and compatibility with clinoptilolite. The plastic pellets, used primarily for injection molding and extrusions, were usually in the form of .316 cm (0.125 in.) diameter cylinders by .316 cm (0.125 in.) long and were comparable to No. 2 Anthrafilt. Even though all the plastics were lighter than Anthrafilt, each formed a zone of integration with the clinoptilolite. However, a resin marketed by General Electric Plastics Division as NORYL proved to be compatible with clinoptilolite, as it formed only a very slight integrated interphase with -20+50 mesh clinoptilolite, and no integrated interphase with -16+30 mesh clinoptilolite.

As a result of this investigation, the pilot runs were conducted using Noryl instead of Anthrafilt as the primary medium and -16+30 mesh clinoptilolite was selected as the secondary medium. The -16+30 clinoptilolite had an effective size of 0.615 mm and a uniformity coefficient of 1.76.

COAGULATION AND FLOCCULATION STUDIES

Polyelectrolyte Selection

A total of 23 polyelectrolytes were evaluated for their ability to produce a dense floc and a clear supernatant. The results from dosing an alum floc with 10 mg/l of various polyelectrolytes are shown in Table 2.

TABLE 2. POLYELECTROLYTE EVALUATION

Polymer	Floc Characteristic	Supernatant (after 15 min. Settling)	Settling Characteristic
Dow Purifloc			
A23	Large Fibrous	Clear	Rapid
C31	Fibrous	Cloudy	Floating Floc
C32	Fibrous	Cloudy	Floating Floc
C41	Fibrous	Clear	Floating Floc
N11	Fibrous	Clear	Rapid
N12	Fibrous	Clear	Rapid
N17	Fluffy	Cloudy	Very Slow
N20	Fibrous	Clear	Slow
Calgon			
CAT-Floc	Fluffy	Cloudy	Very Slow
CAT-Floc 8	Fluffy	Cloudy	Very Slow
CAT-Flow LV	Fluffy	Cloudy	Very Slow
WT 3000	Large Fibrous	Clear	Very Rapid
Nalco 71D09	Fibrous	Clear	Rapid
Magnifloc 905N	Fluffy	Cloudy	Slow

The polyelectrolytes that were evaluated at a dosage of 5 mg/l are listed below:

Atlasep 3A3
Calgon WT 3000
Gendrin
Magnifloc 560C
Magnifloc 836A
Nalcolyte 675H
Nalcolyte 670H
Purifloc N11
R&H C7

At the concentration of 5 mg/l, the criterion for selecting a polymer was based on the resulting floc characteristics and its settling rate, since all of the polymers, except for Magnifloc 560C, resulted in a clear supernatant after a 20 minute settling period. The three polymers with the fastest settling floc, in decreasing order were: Atlasep 3A3, Nalcolyte 71D09 and Calgon WT 3000. However, lowering the final poylmer concentration to 2 mg/l, produced cloudy supernatants in all of the samples except the one containing the Atlasep 3A3.

On the basis of the above information, Atlasep 3A3 was chosen over the other polyelectrolytes.

Alum Feed Concentration

Three concentrations of alum feed were evaluated: 10, 5 and 1%. The flocculation energy of the system was varied by changing the length of coagulation coil and/or the flow rate. Varying the concentration of the alum feed solution appeared to have no effect on the formation of floc particles, nor on the efficiency of phosphorus removal. Similar conclusions were arrived at when analyzing the data concerning the energy of flocculation. No related change in TIP removal or floc formation was noticed in response to a change of the flocculation energy.

This leads one to conclude that there are other more determining factors which influence the efficiency of TIP removal and floc formation during alum addition, coagulation, and flocculation. Alternately, if these parameters do affect coagulation and flocculation, they do not do so in the range of concentrations and values investigated in this study.

CLINOPTILOLITE EVALUATION

Clinoptilolite Selection

Since clinoptilolite's ammonia removal efficiency is dependent on its effective ammonia adsorption capacity, it is important to determine those clinoptilolite deposits having the highest ammonia adsorption capacity. Upon the acquisition of this information, the ammonia exchange system can be optimized. To date many of the studies in the literature,

including this bench and demonstration scale study, have utilized clinoptilolite mined near Hector California. Although the Hector deposit is the only one that is commercially active, there are several known sizable deposits in this country. Therefore, it becomes necessary that work be done to establish information concerning the various clinoptilolite deposits. Preliminary investigations to determine the effective ammonia exchange capacity for various clinoptilolite samples were conducted.

Eight samples of clinoptilolite were obtained from different areas of western United States. The samples were from Grant County, New Mexico; Owyhee County, Idaho; Fremont County, Wyoming; Lander County, Nevada; Washoe County, Nevada; Malheur County, Oregon; Maricopa County, Arizona; and San Bernardino County, California, (Hector).

The samples were sieved to -30 +40 mesh (standard screen scale) and then washed with demineralized water to remove any fine material. After drying at 105°C for 18 hours, the samples were again sieved to a -30 +40 mesh size distribution.

In order to determine an effective exchange capacity for each of the eight samples, static equilibrium tests were performed utilizing an equipoise shaker. Two grams of each sample were placed in a one-liter container to which 500 ml of primary wastewater (previously passed through a 200 mesh screen) was added. In all tests the ammonia-nitrogen concentration of the wastewater was approximately 20 mg/l. The applied wastewater was analyzed for NH_3-N, TKN, K, Ba, Ca, Na, Fe and Mg. The clinoptilolite sample and applied wastewater were shaken on the equipoise shaker for two hours after which a sample of the wastewater in each container was taken and quantitatively analyzed for each of the above parameters. The clinoptilolite was then filtered and contacted with an additional 500 ml of wastewater. A maximum of three repetitions of the above procedure was conducted. The resulting integrated reduction in NH_3-N yielded the effective adsorption capacity for each clinoptilolite sample.

The data from the static tests indicated that a substantial difference exists in the effective exchange capacities exhibited by clinoptilolite samples acquired from the various deposits. Results of the first and second series of tests are presented in Table 3 and 4, respectively. Although influent NH_3-N concentrations did vary slightly from 18 mg/l to 21 mg/l between the two tests, the relationship between the clinoptilolite exchange capacities did not show any variation. The clinoptilolite samples which showed the highest effective exchange capacity were obtained from Malheur County, Owyhee County, and Grant County. Those samples exhibiting the lowest effective exchange capacity were obtained from Maricopa County and Washoe County. The Malheur County sample had an effective exchange capacity which was 170% greater than that determined for the sample obtained from Maricopa County in the first series and over 160% greater in the second series. In the initial exchange capacity evaluations, the Malheur County sample had a measured effective exchange capacity of approximately 0.377 meq NH_3-N/g compared to only 0.140 meq NH_3-N/g of clinoptilolite acquired from Maricipa County deposits. In the second series of evaluations only two runs (two hours each) were made, and

as a result of the reduced contact time, the clinoptilolite samples did not exhibit their optimum effective exchange capacity. However, the higher applied NH_3-N concentration of 21 mg/l resulted in an increase in the exchange capacity of the Malheur County clinoptilolite which was determined to be 0.390 meq/g. The Maricopa County clinoptilolite sample exhibited an exchange capacity of 0.162 meq/g under the higher NH_3-N wastewater concentration.

TABLE 3. EFFECTIVE EXCHANGE CAPACITIES OF CLINOPTILOLITE SAMPLES

Series 1 Concentration NH_3-N applied - 18 mg/l NH_3-N
NH_3-N applied per contact - 9 mg
Total NH_3-N applied - 27 mg

Sample	mg NH_3-N Adsorbed	meq NH_3-N/g Clinoptilolite
Malheur County	10.57	0.377
Grant County	8.05	0.288
Hector Clinoptilolite	8.01	0.286
Owyhee County	6.92	0.247
Lander County	6.31	0.225
Fremont County	5.24	0.187
Maricopa County	3.90	0.139
Washoe County	3.80	0.140

TABLE 4. EFFECTIVE EXCHANGE CAPACITY OF CLINOPTILOLITE SAMPLES

Series 2 Concentration NH_3-N applied - 21 mg/l
NH_3-N applied per contact - 9 mg
Total NH_3-N applied - 27 mg

Sample	mg NH_3-N Adsorbed	meq NH_3-N/g Clinoptilolite
Malheur County	10.92	0.390
Owyhee County	9.61	0.343
Grant County	9.28	0.331
Hector Clinoptilolite	8.72	0.331
Lander County	7.00	0.250
Fremont County	5.25	0.187
Maricopa County	4.55	0.162
Washoe County	4.10	0.146

Regarding the other parameters tested, it was observed that the clinoptilolite samples having the highest effective ammonia exchange capacity exhibited a tendency to desorb sodium while the samples with the lowest capacities adsorbed sodium. One could conclude that the natural occurring clinoptilolites in the sodium form may be the most effective NH_3-N exchangers.

The results indicated that for the eight clinoptilolite samples tested, four, Malheur, Grant, Owyhee, and Hector, had higher effective capacities while those of Maricopa, Washoe, Fremont, and Lander had substantially lower exchange capacities.

Clinoptilolite Activation

Clinoptilolite is rarely found in a pure state when it is mined; there are usually some impurities associated with it. Depending on the origin of the clinoptilolite, the impurities may consist of metal oxides, carbonates or even other zeolites. The presence of these impurities can impair clinoptilolite's ability to remove ammonia from a contacting solution. The removal of other zeolites from clinoptilolite can be very difficult, and costly; however, a number of other impurities can possibly be eliminated by chemical treatment.

The effect of chemical pretreatment was evaluated for five clinoptilolite samples obtained from different deposits located in western United States. The procedure consisted of contacting 2 g samples of clinoptilolite with a 2N solution of hydrochloric acid, (HCl), followed by treatment with a caustic-brine solution (0.5 g/l HaOH + 20 g/l NaCl). Following this treatment, the samples were rinsed with deionized water and then subjected to effective exchange capacity evaluations.

The results from the activation tests are presented in Table 5. In some cases pretreatment increased the effective exchange capacity as was the case in clinoptilolite samples obtained from deposits in Malheur County, Oregon and Maricopa County, Arizona. The most significant increase in the effective exchange capacity was observed with the sample from Maricopa County, Arizona. Its effective exchange capacity almost doubled (0.162 to 0.309 meq NH_3-N per g clinoptilolite) after treatment with caustic-brine solution. The Malheur County sample exhibited an increase of approximately 22 percent (0.390 to 0.475 meq NH_3-N per g clinoptilolite) upon activation. The remaining clinoptilolite samples from deposits contained in Owyhee County, Idaho; Washoe County, Nevada; and Hector clinoptilolite from San Bernandino County, California showed a slight decrease in their effective exchange capacity after the caustic-brine chemical treatment. The most significant decrease in the effective exchange capacity occurred for the Hector clinoptilolite sample which exhibited a decrease from 0.311 to 0.273 meq NH_3-N per g of clinoptilolite. This represents a 12 percent decrease in the effective exchange capacity.

TABLE 5. EFFECT OF PRETREATMENT ON THE EEC OF CLINOPTILOLITE

10.9 mg NH_3-N Per Contact Period
Total NH_3-N Applied: 21.8 mg

DEPOSIT LOCATION	EEC VALUE PRETREATMENT BEFORE	AFTER
Malheur	0.390	0.475
Owyhee	0.343	0.330
Hector	0.311	0.273
Maricopa	0.162	0.309
Washoe	0.146	0.125

PHOSPHORUS REMOVAL

Successful phosphorus removal in the single unit process design was dependent on the combination of the effects of both direct and indirect factors. The direct factors are considered those associated with the precipitation and complexation of phosphorus and its eventual incorporation into an aggregated alum floc, whereas, the indirect factors are associated with the effective destabilization and subsequent removal of the floc. Included in the direct factors are alum dosage, mixing and flocculation energies, pH, temperature and the phosphorus concentrations associated with the influent wastewater. Indirect factors include parameters such as media size, bed depth, rate of filtration and influent suspended solids concentration.

Influent phosphorus concentrations of 4.3 mg/l were effectively and consistently reduced to levels below 0.10 mg/l with the addition of either 100 mg/l or 200 mg/l of alum. This is not surprising since the resultant aluminum to phosphorus (Al:P) molar ratios of 2.1 and 4.3, respectively, are equal to or greater than the Al:P molar ratio of 2.0 that most investigators have found to be required for optimum phosphorus removal. An alum dosage of 50 mg/l was just as effective as the 100 mg/l and 200 mg/l dosages in reducing phosphorus levels below 0.10 mg/l. Based on influent concentrations of 2.6 mg/l and 2.0 mg/l, the resultant Al:P molar ratio is computed to range between 1.7 and 2.3, respectively. Alum dosages of less than 50 mg/l, which would have resulted in even lower Al:P ratios, were not evaluated.

Mixing and flocculation energies are known to influence the effectiveness of alum in removing phosphorus. Since the system under investigation relied entirely on in-line addition and mixing of alum with subsequent in-line flocculation, the energies of mixing and flocculation were particularly important and dependent on flow rate. While the system operated in a decreasing flow mode of operation, the effect of these energies could be readily evaluated by monitoring the effluent phosphorus concentrations at the commencement and termination

of the runs. Effluent phosphorus concentrations at terminal flow rates were usually lower than the effluent phosphorus concentrations at the initial flow rates in spite of a decrease in the mixing and flocculation energies induced by decreased flow rates.

AMMONIA REMOVAL

By analyzing the ammonia removal efficiency under conditions of the various runs, it was possible to verify the effect of system variables such as polymer dosage, alum dosage, primary medium depth and clinoptilolite depth on the ammonia removal process. In addition, three tests were developed to specifically determine the effect of particle size on the ammonia removal capacity of the clinoptilolite.

In determining the ammonia removal capacity, clinoptilolite depths of .38 m (1.25 ft) and .76 m (2.5 ft) were compared. Still another test compared the ammonia removal capacities of 2 different particle sizes. The smaller mesh clinoptilolite used throughout the entire study, had an effective size of .615 mm and a uniformity coefficient of 1.76, while the larger mesh had an effective size of 1.47 mm and a uniformity coefficient of 1.56. These tests were run with no additions of either alum or polymer, in order to preclude the chances of the column plugging up with floc and thereby interrupting the test.

Analysis of the data revealed that the ammonia removal mechanism is unaffected by either alum concentration, polymer concentration or the primary medium depth. Similarly, the saturation of the clinoptilolite layer with floc particles, as is the case during a solids breakthrough condition, does not appear to inhibit the effectiveness of the clinoptilolite as a result of the blinding of surface or pore structure active sites.

The clinoptilolite performed reasonably well under the conditions of a high application rate and a low contact time. In spite of these unusual opeating conditions, initial effluent ammonia concentrations below 0.2 mg/l were consistently recorded. Typical flow rates for exchange-adsorption mechanisms are in the order of 10 bed volumes/hour; however, in the course of this study, depending on the clinoptilolite bed depth, typical flow rates ranged from 30 to 60 bed volumes/hour.

As is evidenced in Figure 1 the ammonia removal efficiency at the termination of a typical test run was 80% or greater in the majority of tests. This indicates that the rate of increase of headloss far exceeds the exhaustion rate of the clinoptilolite in the course of a normal run. This suggests that several backwashings of the primary filter medium may be in order prior to clinoptilolite regeneration. No conclusions could be drawn regarding the effect of clinoptilolite depth on ammonia removal efficiency, because the removal efficiencies for all depths were quite similar, indicating minimal exhaustion as witnessed in Figure 1.

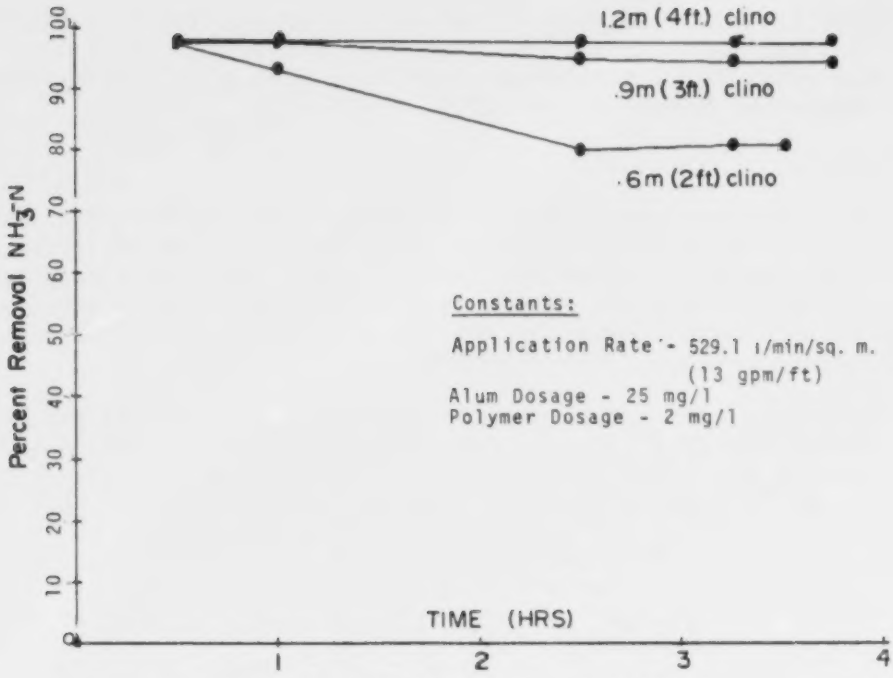

FIGURE 1. AMMONIA REMOVAL EFFICIENCY

To overcome this problem a number of tests were specifically run to determine the effective exchange capacity of the clinoptilolite. These tests indicated that the removal efficiency varied directly with the clinoptilolite bed depth as tabulated in Table 6.

TABLE 6. EFFECT OF CLINOPTILOLITE DEPTH ON AMMONIA REMOVAL EFFICIENCY

Constants: Primary Media Depth = 2.44 m (8.0 ft.)
Application Rate = 530 l/min/m^2 (13 gpm/ft^2)
Alum Dosage = 0.0 mg/l
Polymer Dosage = 0.0

Run Time (Hr.)	1.25 ft Clinoptilolite Throughput Volume liters (Gal.)	NH_3-N (mg/l)	% NH_3-N Removed	2.5 ft Clinoptilolite Throughput Volume liters (Gal.)	NH_3-N (mg/l)	% NH_3-N Removed
0	0 (0)	9.00	0.0	0 (0)	9.00	0.0
.5	65.9 (17.4)	0.76	91.6	67.4 (17.8)	0.19	97.9
1.0	130.2 (34.4)	2.45	72.9	135.5 (35.8)	0.19	97.8
1.5	196.4 (51.9)	3.96	56.1	204.4 (54.0)	0.28	96.9
2.5	317.6 (83.9)	5.71	36.7	351.6 (92.9)	1.27	85.9
3.5	442.1 (116.8)	6.72	24.4	497.0 (131.3)	1.81	68.8
4.5	567.0 (149.8)	7.23	19.8	635.9 (168.0)	4.13	54.1
5.0	629.1 (166.2)	7.38	18.0	702.5 (185.6)	4.64	48.4
5.5	691.9 (182.8)	7.54	16.2	766.1 (202.4)	4.94	45.1

The advantages of this greater clinoptilolite bed depth are readily apparent in Table 6 which summarizes the data. It is evident that at the onset of the first run the effluent NH_3-N concentration is far lower in that for the run using the greater clinoptilolite bed depth. Secondly, a lower effluent NH_3-N concentration is estimated for a greater period of time. For example the volume of wastewater processed, before an effluent NH_3-N concentration of 1.0 mg/l is obtained, is approximately 322 l (85 gal) and 76 l (20 gal) for the .76 and .38 m (2.5 and 1.25 ft) depths of clinoptilolite, respectively. This is more than a fourfold increase in volume throughput for a twofold increase in bed depth. As the tests proceed, however, and both samples become more exhausted, this difference in removal efficiency between the two bed depths narrows. This can be better visualized in Figure 2, which shows the percent ammonia removal for both bed depths converging as the tests proceed.

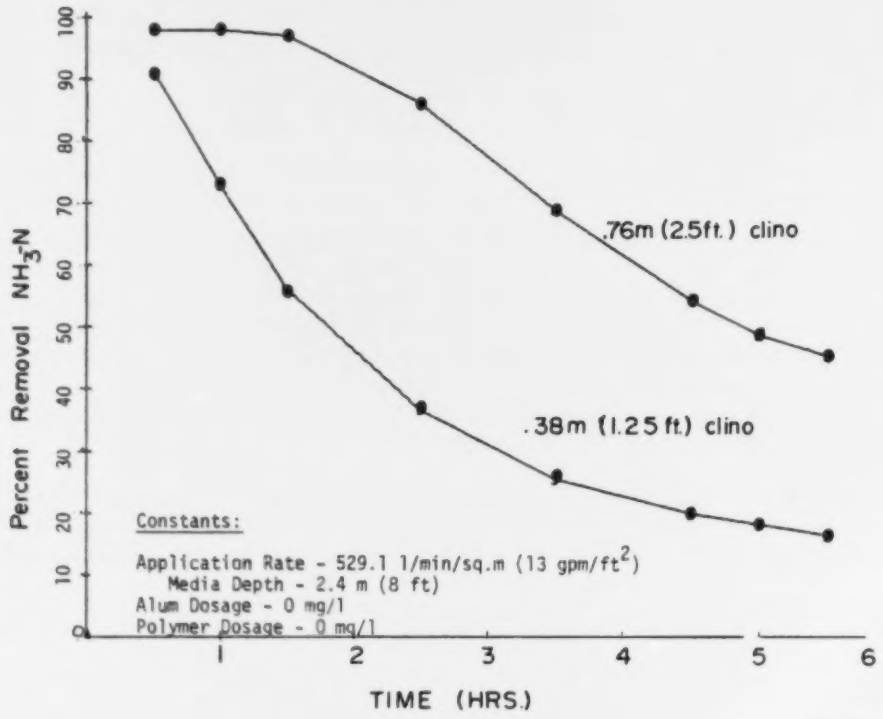

FIGURE 2. EFFECT OF CLINOPTILOLITE DEPTH ON AMMONIA REMOVAL EFFICIENCY

In addition these two tests were used to determine the effective exchange capacity (EEC) of the clinoptilolite, under the conditions of an application rate of 530 l/min/m (13 gpm/ft), an empty bed contact time of approximately 2.5 minutes and an influent NH_3-N concentration of 9.0 mg/l. The EEC was based on a 50 percent reduction of the influent NH_3-N concentration rather than after complete exhaustion of the clinoptilolite. The results of these calculations are presented in Table 7.

TABLE 7. EFFECTIVE EXCHANGE CAPACITY OF CLINOPTILOLITE (EEC) UNDER CONSTANT FLOW CONDITIONS

(Based on 50% NH_3-N Reduction in Influent)

Application Rate: 530 l/min/m^2 (13.6 gpm/ft^2)
Clinoptilolite Size: -16+30 mesh
Effective Size: 0.615 mm
Uniformity Coefficient: 1.76

Clino Depth meters (ft)	Clino Wt. (g)	Throughput Vol. @ 50% Red. (l)	(gal)	Flow (Bed Vol. per hour)	NH_3-N Removed (mg)	Effective Exchange Capacity (meq NH_3-N/g Clino)
.38 (1.25)	1325	227	60	83.6	1590	0.086
.76 (2.50)	2625	681	180	41.8	4990	0.140

It is worth mentioning that typical exchange or adsorption processes proceed at flow rates of 10 to 15 bed volumes per hour. In the above two cases the tests were allowed to proceed at a surface application rate of 530 l/min/m (13 gpm/ft). The flow rates (BV/hr) through the clinoptilolite portion of the system were substantially greater than one would expect, as is shown in Table 7. In comparing the EEC of the two tests it is anticipated that reducing the flow rate through the clinoptilolite media to 10 BV/hr may further increase the EEC.

The result from tests evaluating the effect of clinoptilolite particle size on ammonia removal efficiency showed that the two varied inversely. A graphic representation of the data is presented in Figure 3.

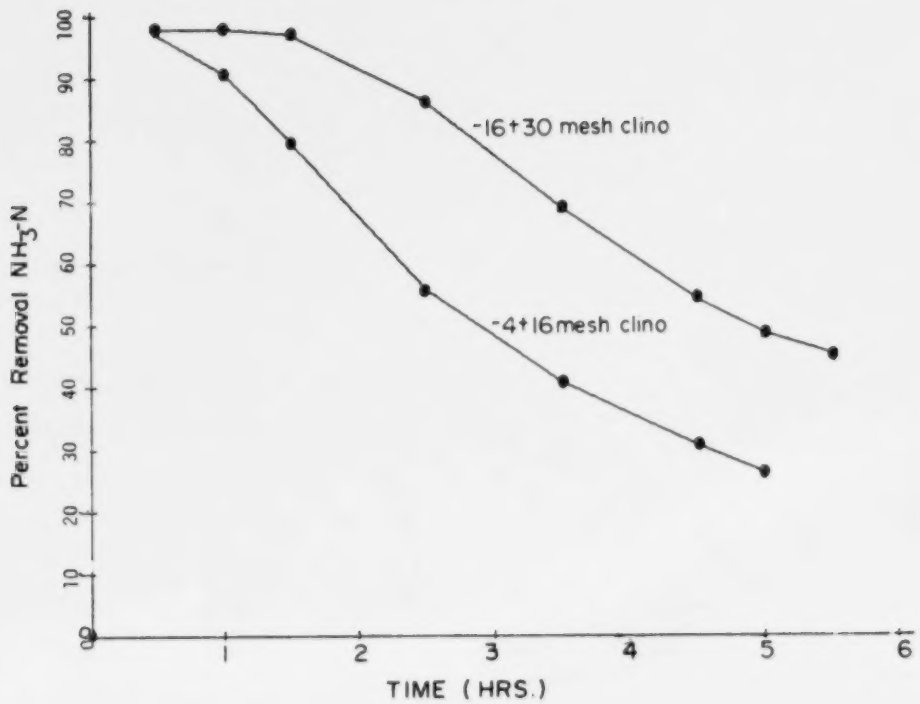

FIGURE 3. EFFECT OF CLINOPTILOLITE PARTICLE SIZE ON AMMONIA REMOVAL EFFICIENCY

Although the run with the larger clinoptilolite size had a one foot depth advantage over the smaller clinoptilolite size, it nevertheless had 25-30 percent lower ammonia removal capability throughout the course of the run. Figure 4 illustrates the significance of particle size in an exchange or removal process. In spite of the fact that the contactor volume represented by the larger particle size clinoptilolite was almost three times the volume of that used in testing the smaller size clinoptilolite, its removal efficiency was inferior, which subsequently led to its quicker exhaustion.

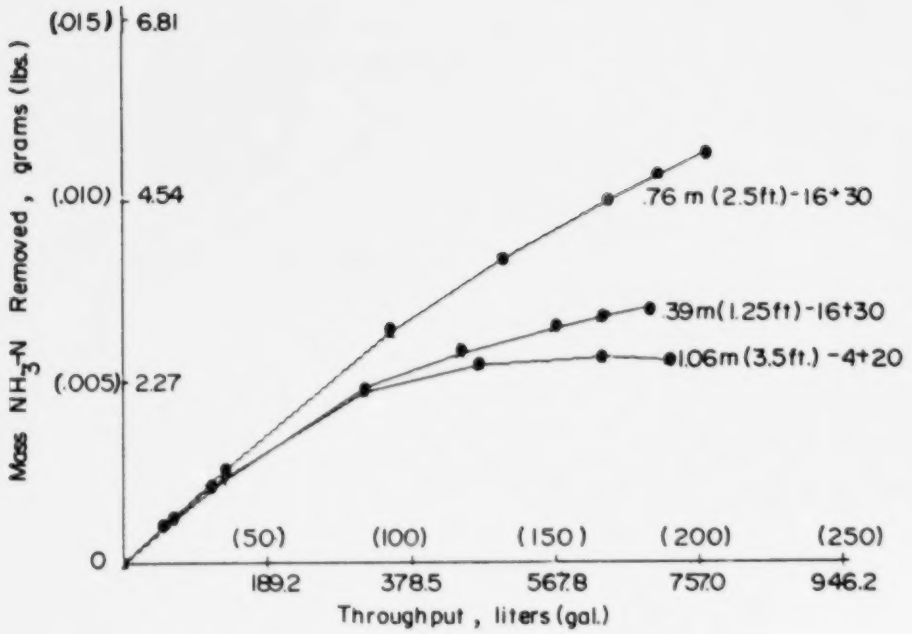

FIGURE 4. EFFECT OF CLINOPTILOLITE PARTICLE SIZE AND DEPTH ON AMMONIA REMOVAL EFFICIENCY

This indicates that a method of increasing the efficiency of ammonia removal by clinoptilolite would be to utilize finer mesh sizes. However, there is a point of diminishing returns. The finer the selected clinoptilolite mesh size, the greater the rate of headloss buildup exhibited across the column. Therefore, one has to weigh the merits of decreased frequency of column regeneration against the increased rate of backwashing and subsequent pumping requirements.

SECTION 5

DEMONSTRATION PILOT PLANT EXPERIMENTAL DESIGN AND SYSTEM DESCRIPTION

DESIGN PHILOSOPHY

General

Analysis of combined sewer overflows and creek water indicated that both the ammonia and phosphorus influent concentrations ranged from approximately 0.2 - 0.5 mg/l. In order to appraise the wider range of concentration that could exist, the wastewaters were spiked with dibasic ammonium phosphate, $(NH4)_2HPO4$. The ammonium phosphate solution was pumped into the influent holding tank; the turbulence in the tank was sufficient to provide adequate mixing of the spiking solution and the wastewater. A variable speed pump delivered the volume of solution necessary to obtain the desired 7 to 15 mg/l concentration of ammonia in the applied wastewater. The phosphorus concentration was predetermined by the desired ammonia concentration.

Four initial flow application rates were evaluated during the study: 529, 407, 326, and 65 l/min/m^2 (13, 10, 8 and 1.6 gpm/ft^2). The combined sewer overflow was applied to the system at two initial rates, 529 and 407 l/min/m^2 (13 and 10 gpm/ft^2) whereas, the creek water was applied at all four application rates. Application rates of 326 and 65 l/min/m^2(8 and 1.6 gpm/ft^2) were only used in tests evaluating ammonia nitrogen removal with clinoptilolite. The columns were operated in the declining rate filtration mode.

The filter was always operated in the downflow mode. On the other hand, the clinoptilolite columns were operated in the upflow mode, except for two creek water runs which were operated in a downflow mode.

Solids Removal

High rate filtration, 407 l/min/m^2 (10 gpm/ft^2), was used to remove the suspended solids from the combined sewer overflows. A previous study reported suspended solids removals from 40 to 60 percent when using high rate filtration (8). However, in this study high concentrations of suspended solids in the effluent from the filter could adversely affect the total system performance by plugging the clinoptilolite columns, particularly if the clinoptilolite columns were operated in the downflow mode. Therefore, high suspended solids removal efficiencies were desired. Following the addition of alum, an anionic polymer (Magnifloc 836A) was added to increase the strength of the alum floc so that a high degree of removal could be obtained. The filter media was an anthracite material with an effective

size of 6.1 mm and a uniformity coefficient of 1.3.

Both the alum and polymer were stored as a 25 percent and 0.04 percent by weight solutions, respectively, and then pumped into the treatment process influent stream at a rate to yield the desired dosages. Alum dosages used during the study ranged from 50-220 mg/l; the polymer dosages ranged from 1 - 4 mg/l. The effect of various alum and polymer dosages on suspended solids removal, as well as, the wastewater application rate were evaluated.

Phosphorus Removal

Phosphorus removal from the combined sewer overflow was achieved in a two-step process; first, precipitation of the phosphorus, and second, removal of the phosphate floc particles from the waste stream by high rate filtration. Alum was added to the wastewater to precipitate the phosphorus. The chemistry of phosphorus removal with alum has been presented in detail elsewhere (9). Two equations used to illustrate the reactions which occur when alum is added to phosphorus-containing wastewater are as follows:

(1) $Al_2(SO_4)_3 \cdot 14\ H_2O + 2\ PO_4^{-3} = AlPO_4 + 3\ SO_4^{-2} + 14\ H_2O$

(2) $Al_2(SO_4)_3 \cdot 14\ H_2O + 6\ HCO_3^- = 2\ Al(OH)_3 + 6\ CO_2 + 14\ H_2O + 3\ SO_4^{-2}$

In the first equation an aluminum phosphate precipitate is formed; actually the precipitate is probably of an intermediate composition between the crystalline solid $AlPO_4$, $Al(OH)_3$ (second reaction) and an amorphous structure. From the first reaction the aluminum to phosphorus (Al:P) molar ratio appears to be 1:1; however, the second reaction competes with the phosphorus for the aluminum, and is thus at least partially responsible for the excess alum which is required. With the addition of alum the second reaction results in the loss of alkalinity, with the subsequent pH reduction dependent on the initial alkalinity concentration.

The solubility of the $AlPO_4$ is pH dependent. Theoretical solubility concentrations have been calculated at various pHs. The calculations have shown that the minimum solubility of $AlPO_4$ occurs in the pH range of 5.5 - 6.5.

Since the combined sewer overflows were spiked with dibasic ammonium phosphate, $(NH_4)_2\ HPO_4$, essentially all the phosphorus in the phosphorus in the wastewater was in the soluble form, as an orthophosphate PO_4^{-3}

The process variables normally optimized in phosphorus removal evaluations include the applied alum concentration, and the resultant Al:P molar ratio. Phosphorus removal with alum addition has been quite extensively studied; however, in most of those studies the alum and phosphorus-containing wastewater were contacted in rapid-mix tanks for several minutes and the aluminum phosphate precipitate subsequently removed by sedimentation in a settling tank. Since the system evaluated in the course of this study consisted of in-line static mixers and filtration columns, the physical processes were substantially different from those evaluated by other investigators, and

the process determining variables, stated above, normally associated with phosphorus removal may not be determinimg in this study.

The contact time in the in-line static mixer ranged from 0.4 to 1.2 seconds compared to several minutes in the rapid-mix tank, and the time in the filter ranged from 2 - 4 minutes compared to 60 to 120 minutes in the settling tank. Also, as the particular run proceeded, the filtration column simulated a packed bed reactor, containing alum which was still available to react with the phosphorus. Jenkins, et al. (9) indicated that the initially formed precipitate tended to contain an excess of hydroxide over the phosphate solid, thereby increasing the possibility that the recycle and reuse of precipitated solids would be effective in increasing phosphate removal. Therefore, the presence of an alum sludge in the filter may remove additional phosphorus. If this proves to be the case, then once alum sludge is accumulated, the required Al:P molar ratio may be decreased due to the additional available alum contained in the sludge.

As previously mentioned, the calculated velocity gradients in the static mixers, 6085 fps per foot at 529 $l/min/m^2$ (13 gpm/ft^2) application rate, and 1390 fps per foot at 203 $l/min/m^2$ (5 gpm/ft^2) application rate, were much higher than the velocity gradients normally associated with conventional rapid-mix tanks. The data was analyzed to determine if the decreasing flow rate had any effect on the phosphorus removal, possibly related either to decreasing velocity gradients or increasing contact time in both the mixer and filtration column.

The alum dosages evaluated included 50, 100 and 200 mg/l. Operating under a declining rate filtration necessitated constant modification of the alum flow rate. This caused minor variations in the alum dosage during a particular run. Since the alum flow rates were monitored, the actual alum dosages were recorded and Al:P molar ratios were established.

Ammonia Removal

The natural zeolite, clinoptilolite, was used as an ion exchange agent to remove ammonia from the combined sewer overflows. Previous studies by others using clinoptilolite with municipal, industrial, and synthetic wastewaters have revealed many of the variables which effect the adsorption capacity of the clinoptilolite (6,10). Variables such as wastewater pH, cationic strength, nature of the competing cations, ionic form of the zeolite (H^+, Na^+, or Ca^{++}), grain size of the clinoptilolite, influent ammonia concentration, and contact time, have all been shown to effect the adsorption capacity of the clinoptilolite.

Only the influent ammonia nitrogen concentration and contact time were purposely varied to determine their impacts on the EEC of clinoptilolite and the effluent quality. It was realized that the competing ion concentrations would vary somewhat throughout the testing; however, this variation was not expected to be significant enough to warrant the monitoring of competing cations. Other factors such as the variation in the influent ammonia levels were felt to be much more significant. In fact Koon, et al, (6) have stated that the impact of the cationic strength will vary for

different combinations of cations.

The clinoptilolite was received in the form of a -16 +50 mesh size (<1.19 mm(0.047 in.); > 0.3 mm (0.012 in)), with about 20% finer material. The clinoptilolite columns were backwashed at 582 l/min/m^2 (14.3 gpm/ft^2) to remove the finer material, before each series of tests. Samples were collected from the surface, 0.30 m (1 ft) and 0.91 m (3 ft) depths and a sieve analysis was conducted to determine the size distribution of the clinoptilolite after backwashing. Table 8 illustrates the particle size distribution pattern after backwashing. Prior to backwashing the clinoptilolite had an effective size of 0.42 mm and a uniformity coefficient of 1.40.

TABLE 8. CLINOPTILOLITE PARTICLE SIZE DISTRIBUTION AFTER BACKWASHING

DEPTH (ft.)	EFFECTIVE SIZE (mm)
Surface	0.35
1	0.42
3	0.59

Several influent NH_3(N) concentrations were used in the various testing series. Although throughout a particular testing run an attempt was made to keep the influent NH_3(N) concentration fairly constant, certain variations in the influent did occur; therefore, the weighted average influent concentration was used for comparison between test series.

With the exception of two cases, all the runs proceeded with the clinoptilolite columns operating in the upflow mode at flow rates already mentioned. Although suspended solids may not have interfered with the actual adsorption capacity of the clinoptilolite, the possibility that breakthrough of the polymer-flocculated solids would plug the finer mesh clinoptilolite columns necessitated operating those columns in the upflow mode.

Tests were conducted to determine the effect of particle contact time on the adsorption capacity of the clinoptilolite. The particle contact time was assumed to be different from the empty bed residence time. The empty bed residence time can be altered by increasing the bed depth. The particle contact time is defined as the time that the dissolved NH_3(N) molecule is in contact with an individual clinoptilolite particle and therefore is dependent on the wastewater velocity through the bed and independent of bed depth.

Although the actual particle contact time would have been extremely difficult to calculate, it was thought that a five-fold difference in the velocity of the wastewater passing through two columns of the same clinoptilolite depth could illustrate the dependence of the contact time on the EEC of the clinoptilolite.

The first 13 individual runs were operated at an influx of 407

l/min/m^2 (10 gpm/ft^2) and following alum and polymer addition the wastewaters were applied in an upflow manner to a column containing 1.52 m (5 ft) of clinoptilolite. The second series of tests consisted of 12 individual runs operating in an upflow mode through the clinoptilolite and at an influx of 529 l/min/m^2 (13 gpm/ft^2). However, the second series operated with two clinoptilolite columns each filled with 1.37 m (4.5 ft) of clinoptilolite and connected in series. A media height of 1.37 m (4.5 ft) in the clinoptilolite columns was limiting, since the columns were operated in the upflow mode with the bed expanding to approximately 2.44 m (8 ft) at the 529 l/min/m^2 (13 gpm/ft^2) influx. Operating two clinoptilolite columns in series allowed a determination of the effective ammonia exchange capacity from the data derived from the first column as well as the effectiveness of the series configuration.

The third series of tests involved the application of creek water to the clinoptilolite column in the downflow mode. Since the creek water contained less than 40 mg/l of suspended solids, no alum or polymer were added, thereby eliminating the plugging of the clinoptilolite column. The application influxes were 326 l/min/m^2 (8 gpm/ft^2) and 65 l/min/m^2 (1.6 gpm/ft^2). Since these tests were conducted for the purpose of evaluating the effect of contact time on the effective ammonia exchange capacity, only one clinoptilolite column was used in each test.

Sampling and Analyses

Sampling of the system was conducted both manually and automatically. The filter influent wastewater was sampled in the Zurn screening unit effluent tank using an automatic sampler.

The effluent was sampled with an automatic sampler at the effluent pipe from the clinoptilolite column. A third sampling site, between the two clinoptilolite columns, was introduced when the series configuration utilized two clinoptilolite columns.

Analysis of the influent and effluent wastewaters included: suspended solids (SS), ortho and hydrolyzable phosphate (TIP), ammonia nitrogen (NH_3-N) total alkalinity (TALK), and pH.

The TIP, NH_3-N, and TALK analyses were done utilizing a Technicon Auto Analyzer, while the SS and pH analyses were conducted in accordance with Standard Methods for Examination of Water and Wastewater, 13th Edition.

EQUIPMENT DESCRIPTION

The nutrient removal demonstration facilities were housed in an existing combined sewer overflow treatment facility located at Maltbie Street on the west side of Syracuse, N.Y. Combined sewer overflows from a drainage area of 115 acres zoned for commercial and light industrial use were treated at the Maltbie Street Facility.

The fundamental purpose of the Maltbie Street treatment facility was

to evaluate high rate solids removal and high rate disinfection of combined sewer overflows. The facility was equipped with a pumping station that collected, metered and pumped the wastewater to the treatment section which consisted of three screening units each with a different size screen aperture and followed by a disinfection unit. The screening units were connected in parallel and could be operated independently. Following the screening operation, the wastewater proceeded to the disinfection chambers where it was treated with either chlorine (Cl_2) or chlorine dioxide (ClO_2) and then discharged into the receiving stream, Onondaga Creek. A diagram of the Maltbie Street site is shown in Figure 5.

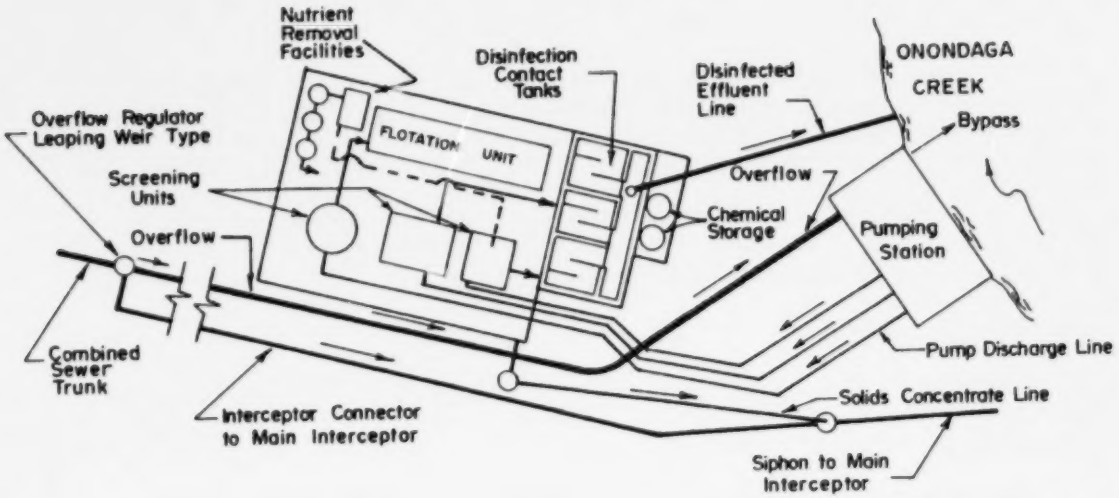

FIGURE 5. MALTBIE STREET SITE PLAN

The combined sewer overflow used in the nutrient removal system was passed through a Zurn MicroMatic horizontal drum screen which had a screen aperture size of 71 microns. After screening, a portion of the flow was pumped to the nutrient removal system. The remainder of the flow proceeded to the disinfection chambers and was subsequently discharged to the creek.

During periods of dry weather, creek water was used as the influent wastewater to the nutrient removal system. Creek water was pumped up to the treatment system by a Barnes submersible pump, Model No. 12 ASE 1, which was capable of pumping 6.31 l/sec (100 gpm) at 15.2 m (50 ft) head. Although the creek water typically had a suspended solids concentration of less than 100 mg/l, it was nevertheless screened through the Zurn unit. After screening it was applied to the nutrient removal system in the same manner as the combined sewer overflow.

The nutrient removal system process flow diagram is presented in Figure 6.

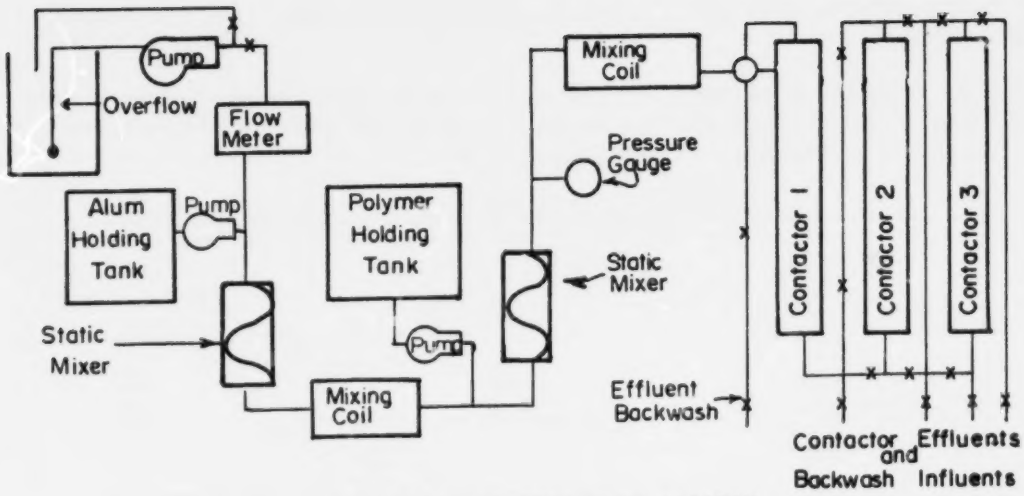

FIGURE 6. PROCESS FLOW DIAGRAM

The screened effluent of the Zurn unit was pumped to the nutrient removal facilities via a centrifugal pump manufactured by Weinman, Model No. 6AE20S having a capacity of 4.4 l/sec (70 gpm) at 19.8 m (65 ft) of head. The end of the intake line of the pump was fitted with a check valve and was placed about 0.61 m (2 ft) from the bottom of the Zurn tank. The discharge line was equipped with a bypass valve that allowed any portion of the pumped CSO to be returned to the Zurn unit. The bypass valve not only allowed for control over the treatment rate but it also provided additional turbulence in the Zurn unit which, to some extent, prevented the solids from settling.

Once the treatment rate was established for a particular run, no further adjustments were made. The system was operated in a declining rate filtration mode.

Flow rate was monitored by a Wallace and Tiernan Rotameter, tube #WT2-40-610 equipped with a stainless steel float #26-12-VI. The effective range of the rotameter was between 0.32 l/sec (5 gpm) and 4.23 l/sec (67 gpm).

Chemical addition of both alum and polymer was accomplished with a duplex head chemical feed pump, manufactured by Madden, Model JR-4. Each side of the feed pump could be adjusted independently so that the proper dosages of each chemical could be added to solution. Alum was added first to the CSO. High energy mixing was achieved with a 5.1 cm (2 in. PVC) in-line static mixer manufactured by Kenics, Model No. 2-50-541-7. The mixer was 0.53 m (1.7 ft) long and contained three standard helixes. Following the high energy mix, a low energy mix, to facilitate flocculation. was provided by installing a 30.5 m (100 ft) coil of 5.1 cm (2 in.) black polyethylene pipe. Polymer addition followed the alum

low energy mixing coil and its mixing sequence was identical to that of alum.

The velocity gradients for the static mixer and the mixing coil were determined at several flux rates for both the high and low energy mixers and are presented in the table below.

TABLE 9. VELOCITY GRADIENTS FOR HIGH AND LOW ENERGY MIXERS

Flow Rate	Velocity Gradient Static Mixer (fps/ft)	Velocity Gradient Mixing Coil (fps/ft)
2.52 l/sec (40 gpm)	3960	572
0.94 l/sec (15 gpm)	1390	140

The above data indicate that velocity gradients of the static mixer, even at terminal flux rates, are considerably higher than those normally associated with rapid mix tanks, 700 fps per foot. With respect to the mixing coils, the data suggested that the coils provided a fluid regime somewhere between mixing and flocculation.

Upon leaving the polymer mixing coil, the wastewater was applied to a filter and subsequently to two ion exchange columns containing clinoptilolite. The filter contained 1.52 to 2.13 m (5 to 7 ft) of No. 3 Anthrafilt. The particle sizes of No. 3 Anthrafilt which is manufactured by Blue Coal Corporation, ranged from 4.76 to 7.92 mm (0.188 to 0.313 in.) and the effective size was determined to be 6.1 mm. Clinoptilolite was purchased from Bariod, Division of National Lead, which owns the Hector clinoptilolite deposits in San Bernardino County, California. The clinoptilolite was processed by the Colorado School of Mines Research Institute at Golden, Colorado, to yield a -16 + 50 mesh material. A sieve analysis of the clinoptilolite as received indicated an effective size of 0.42 mm and a uniformity coefficient of 1.40.

The filter, and ion exchange columns were constructed of 7.11 cm (0.280 in.) plate steel and were 3.05 m (10 ft) by 0.61 m (2 ft) in diameter. Each of the three columns were equipped with 76.2 mm (3 in.) influent and effluent ports. Additionally, the columns were fitted with a 127 mm (5 in.) flanged opening near the top of the column for media loading and a 0.305 m (12 in.) flanged manway near the bottom. Each column contained a perforated steel plate which was used as a media support. Perforations were twelve 38.10 mm (1.5 in.) holes, eight equally spaced on an 20.32 cm (8 in.) radius and four equally spaced on a 10.16 cm (4 in.) radius. The holes in the base plate were fitted with a AISI Type 304 stainless steel Microwedge strainer, manufactured by Roberts Filter Manufacturing Company. The strainers, shown in Figure 7, which had an opening of 0.38 mm (0.015 in.), were approximately 30.2 mm (1.188 in.) in height and 53.98 mm (2.215 in.) diameter and were held in place by a toggle bolt assembly. Since the total area of the base plate was 0.29 m^2

(3.14 ft^2) there was one strainer per 0.024 m^2 (0.26 ft^2).

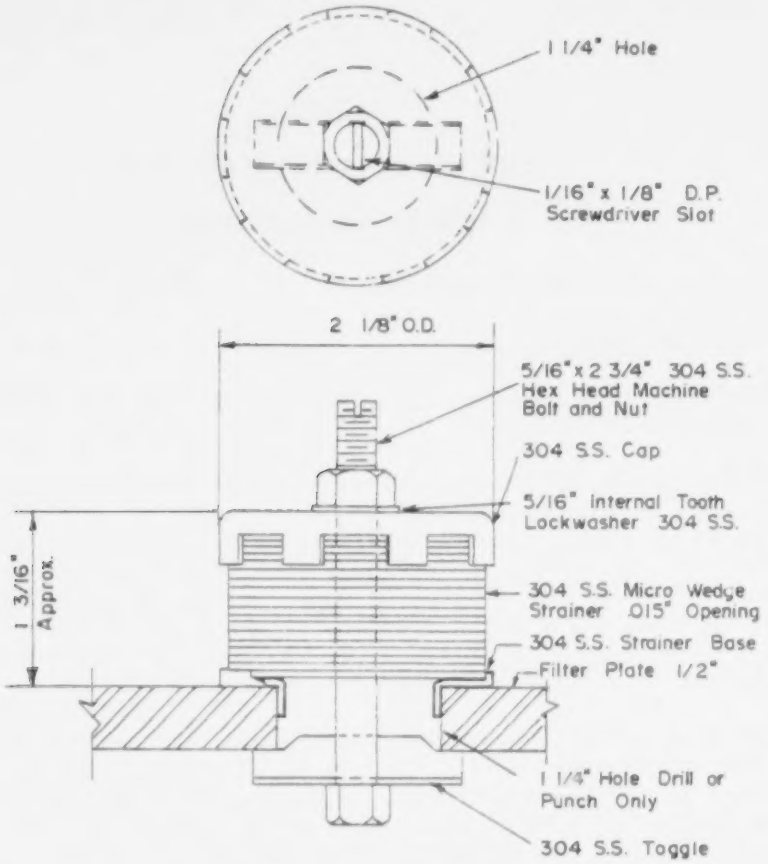
1 1/4" Hole
1/16" x 1/8" D.P. Screwdriver Slot
2 1/8" O.D.
5/16" x 2 3/4" 304 S.S. Hex Head Machine Bolt and Nut
304 S.S. Cap
5/16" Internal Tooth Lockwasher 304 S.S.
1 3/16" Approx.
304 S.S. Micro Wedge Strainer .015" Opening
304 S.S. Strainer Base
Filter Plate 1/2"
1 1/4" Hole Drill or Punch Only
304 S.S. Toggle

FIGURE 7. MICROWEDGE STRAINER ASSEMBLY FOR FILTERS
(Courtesy of Roberts Filter Co.)

The interiors of the columns were sandblasted and coated with two coats 0.25 mm (0.01 in.) of an epoxy enamel (Mobil Hi-Build 78 Series). The epoxy coating was specified since it would withstand the abrasion of the Anthrafilt and clinoptilolite during backwashing. Once the exterior surfaces of the columns were coated with two coats of high gloss enamel, they were placed into position. The columns were connected such that they could be operated in series or parallel. The effluent line of each column was equipped with quick disconnect coupler which allowed for rapid conversion to an upflow mode by connecting two columns with a section of 76.2 mm (3 in.) flexible hose. Backwashing was achieved by connecting a flexible hose from a metered tap water outlet to the quick disconnect coupler at the bottom of each column. Only one column could be backwashed at a time. Similarly, each column was equipped with a quick disconnect coupler for air scouring.

SECTION 6

DEMONSTRATION PILOT PLANT STUDY

RESULTS AND DISCUSSION

COMBINED SEWER OVERFLOW APPLICATION

Suspended Solids Removal

The results of the suspended solids removal data are shown in Table 10 and Figures 8 through 12.

Gross suspended solids removal was accomplished as the wastewater passed through a 73 micron Zurn Micro-Matic screening unit. However, during the first flush, that is the first half hour to hour of a combined sewer overflow occurrence, the effluent from that screening unit contained SS concentrations as high as 250-400 mg/l. Since the system was not automated to start whenever an overflow occurred, many of the runs did not start in time to remove first flush SS. Influent SS concentrations during the first hour of filtration varied from 90 to 250 mg/l.

Since alum addition generates suspended matter and the filter run lengths are dependent on solids loading to the filter, the alum dosages would affect the filter run length. However, the variability in the SS concentration in the applied wastewater prior to chemical addition makes the determination of the effect of alum dosages on a filter run length difficult.

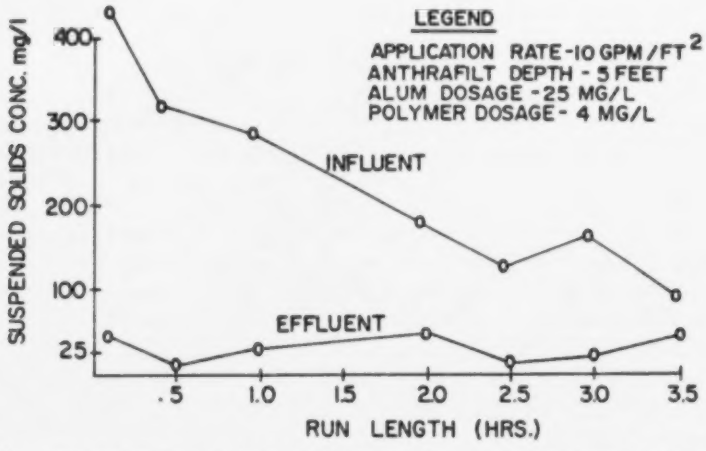

FIGURE 8. SUSPENDED SOLIDS REMOVAL RUN #2 (COMBINED SEWER OVERFLOW)

TABLE 10. SELECTED DATA FOR COMBINED SEWER OVERFLOW RUNS

Run No.	Influx l/min/m² (gpm/ft²)	Final Flux l/min/m² (gpm/ft²)	Run Length (Hrs.)	Alum Dosage (mg/l)	Polymer Dosage (mg/l)	Inf. SS Range (mg/l)	Eff. SS Range (mg/l)	% Removal Range (mg/l	Applied SS Kg (lb)	Residual SS Kg (lb)	% Mass Removed
2a	407 (10)	260.5 (6.4)	3.0	50	4	10- 62	4- 60	67-97			
2	407 (10)	358.1 (8.8)	3.5	50	4	100-435	8- 40	0-99	4.7 (10.36)	.52 (1.61)	89
5	407 (10)	203.5 (5.0)	3.0	100	2	40-260	6- 16	74-95	1.91 (4.21)	.17 (0.39)	91
6	407 (10)	264.5 (6.5)	3.0	100	2	20- 80	15- 25	0-75	.83 (1.84)	.35 (0.77)	58
8	407 (10)	284.9 (7.0)	3.0	100	2	100-288	62-182	34-69	2.98 (6.56)	1.52 (3.35)	49
9	407 (10)	329.6 (8.1)	3.5	50	1.5	43-156	11- 68	56-81	1.49 (3.29)	.5 (1.10)	66
10	407 (10)	215.7 (5.3)	3.5	100	2	22- 63	2- 23	60-85	.62 (1.37)	.19 (0.42)	69
11	407 (10)	382.6 (9.4)	3.0	50	2	13- 69	18- 30	0-82	1.0 (2.20)	.57 (1.27)	43
15	407 (10)	370.3 (9.1)	2.0	50	2	28- 68	7- 47	0-75	.54 (1.19)	.39 (0.87)	28
22	529.1 (13)	293 (7.2)	3.5	215	1	26- 64	0- 10	81-93	1.18 (2.61)	.13 (0.29)	89
23*	529.1 (13)	362.2 (8.9)	2.25	215	1	44- 52	0- 10	85-99	1.03 (2.27)	.10 (0.228)	90
28	529.1 (13)	337.8 (8.3)	3.5	70	2	40-100	7- 30	38-92	1.55 (3.41)	. 4 (0.88)	74
35	407 (10)	284.9 (7.0)	3.0	215	1	100-300	0-150	20-99	3.22 (7.10)	.78 (1.72)	76
36	407 (10)	272.7 (6.7)	3.0	215	1	40-150	2- 60	50-96	1.50 (3.32)	.3 (0.66)	80

*Run Terminated Due to Pump Failure

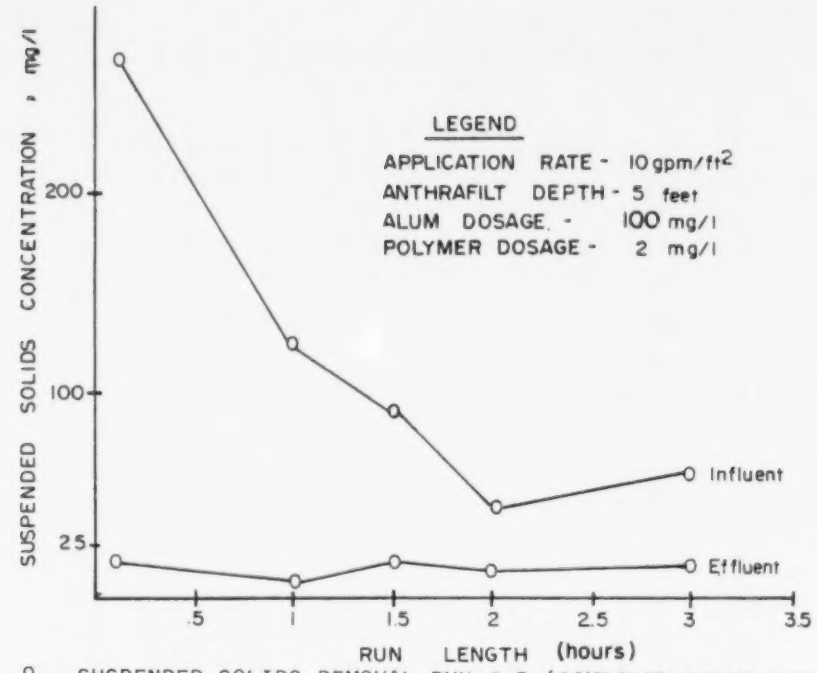

FIGURE 9. SUSPENDED SOLIDS REMOVAL RUN # 5 (COMBINED SEWER OVERFLOW)

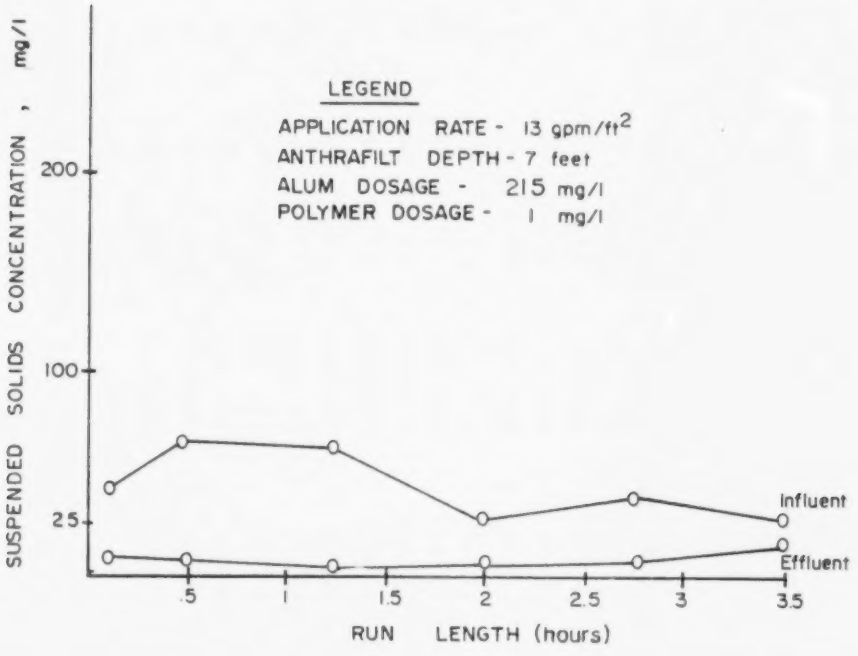

FIGURE 10. SUSPENDED SOLIDS REMOVAL RUN #22 (COMBINED SEWER OVERFLOW)

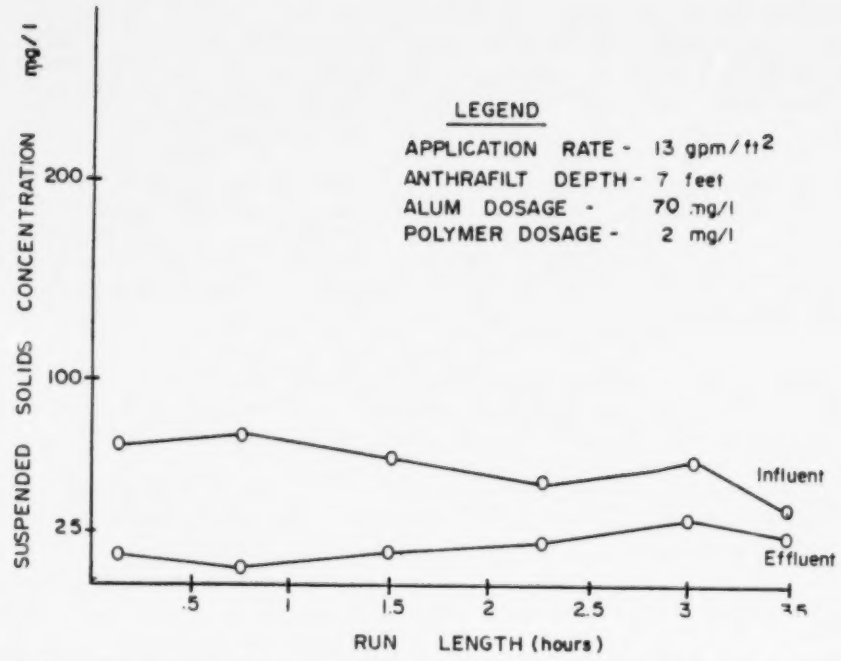

FIGURE 11. SUSPENDED SOLIDS REMOVAL RUN #28 (COMBINED SEWER OVERFLOW)

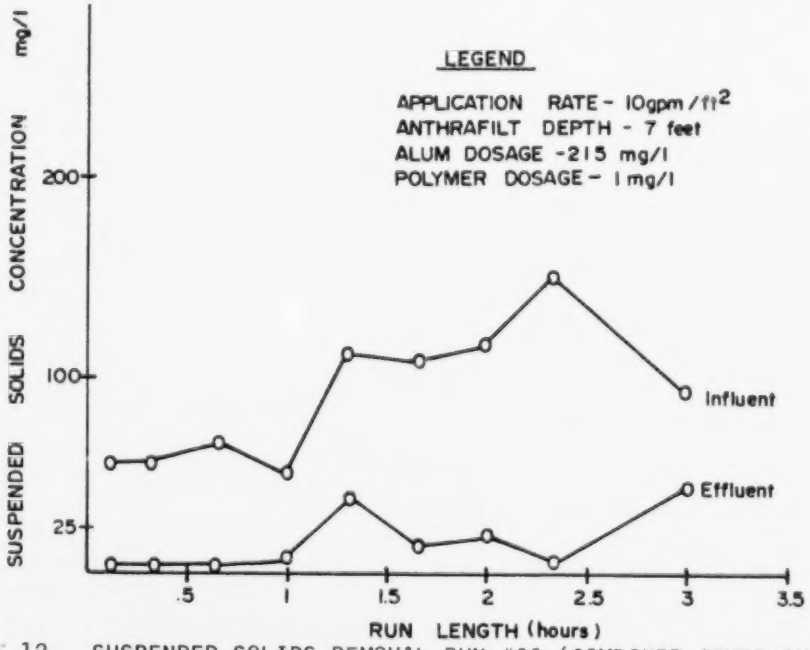

FIGURE 12. SUSPENDED SOLIDS REMOVAL RUN #36 (COMBINED SEWER OVERFLOW)

The run lengths were normally 3 to 3.5 hours, and were terminated when either the total head loss reached 10-13 psi (0.70-0.91 kg/cm^2) or there was a continuous condition of SS breakthrough in the effluent. In cases where the termination was caused by excessive head loss, the terminal flow rates were approximately one to two thirds of the initial flow rates.

Effluent quality from the high rate filter was, for the most part, below 40 mg/l TSS. Since the SS concentration in the CSO varied widely, the removal efficiencies fluctuated with the influent SS concentration. It was especially difficult to obtain high SS removal efficiencies when the influent SS were very low, 20-50 mg/l. Generally, the effluent SS ranged from 20-40 mg/l over the entire filtration run.

It can be seen in Table 10 under the column heading SS Specific Removal, that there was no apparent benefit in operating the filtration column at a depth of 2.14 m (7 ft) as compared to 1.52 m (5 ft).

During most of the runs approximately 15 to 19 m^3 (4000-5000 gal) of CSO were filtered in run lengths of 3-3.5 hours. The total mass of SS removed ranged from 27 to 91% of the total mass of SS applied.

Combined Effect of Alum Dosage and Influent SS--

Figures 13 and 14 best illustrate the effect of the alum and polymer dosages on filtration. Because of data scatter, no well defined optimum dosage could be determined. However, there was a discernible trend

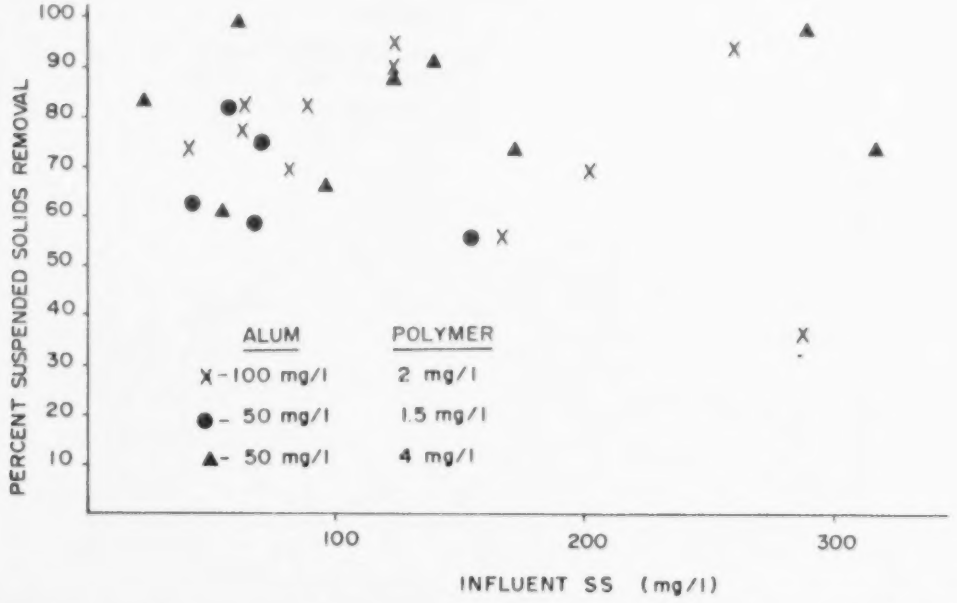

FIGURE 13. PERCENT SUSPENDED SOLIDS REMOVAL VS INFLUENT SUSPENDED SOLIDS

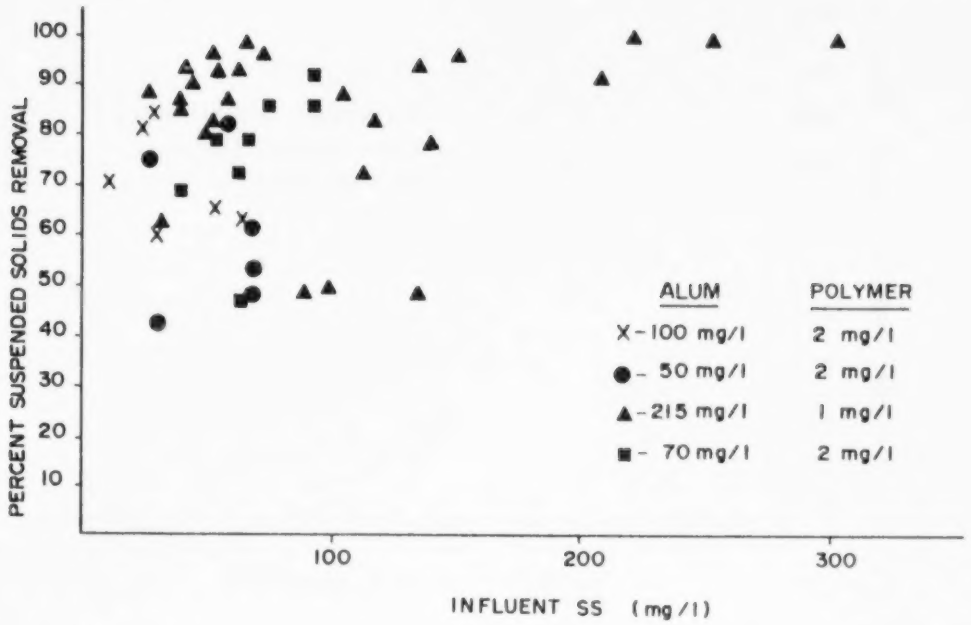

FIGURE 14. PERCENT SUSPENDED SOLIDS REMOVAL VS INFLUENT SUSPENDED SOLIDS

indicating that the lower alum dosages, 50, 70, and 100 mg/l yielded lower quality effluent, ranging from 45 to 70% SS removal. On the other hand, an alum dosage of 215 mg/l appeared to be very effective in removing solids over the entire range of influent SS, 25-400 mg/l, and achieved SS removals between 85 to 99%. This is in agreement with Packham (11) who reported that the amount of alum required for coagulation increased as the turbidity and suspended solids concentration in the influent decreased. This reflects the viewpoint that coagulation improves as the number of suspended particles, which act as nuclei, is increased.

Figure 15 shows the effect of the flow on the SS removal under an applied alum dosage of 215 mg/l, which obtained the best SS removal efficiency. The flow rates are the actual flowrates at the instant a sample was obtained during the runs. This data would tend to indicate that the solids removal system was least efficient at the lower flow rates. However, since the system was operated in the declining rate filtration mode, the lower flow rates are realized at the end of the run when solids breakthrough was most probably occurring. Therefore, it would appear that the flow rate is a parameter which must be closely scrutinized when evaluating the declining rate filtration system.

Phosphorus Removal

The first series of tests were operated under declining rate, starting at 407 l/min/m^2 (10 gpm/ft^2) and decreasing to 203 l/min/m^2

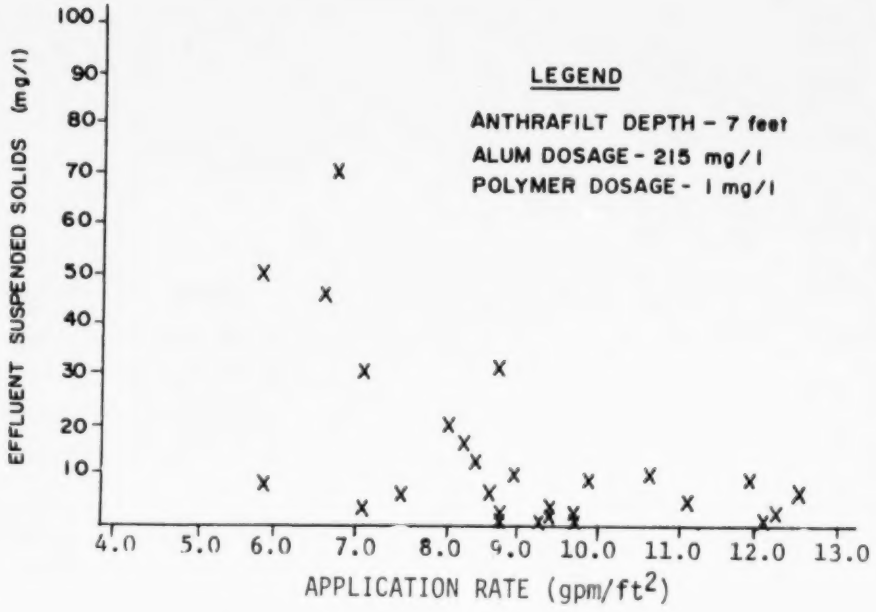

FIGURE 15. EFFLUENT SS Vs APPLICATION RATE

(5 gpm/ft^2) at the termination of the run. The influent phosphorus concentrations ranged from 2.5 to 22 mg/l TIP during this test series. Since a synthetic phosphorus solution was added to the combined sewer overflow, essentially all the phosphorus was in the form of ortho phosphate. Alum dosages of 50 and 100 mg/l used during this series, gave an Al:P molar ratio ranging from 0.3 to 3.9.

Although the data were scattered, the general trend indicated that 95% TIP removal could be obtained with an Al:P molar ratio equal to or greater than 2.0 (Figure 16). With an Al:P molar ratio of 1.0 or greater, the system was assured of at least a 80% TIP reduction. The data showed that alum dosages resulting in Al:P molar ratios less than 1.0 caused the phosphorus removal efficiency to decrease rapidly. It can be seen in Figure 16 that at an Al:P molar ratio of 0.7 only about 75% TIP removal can be expected. Al:P molar ratios lower than 0.7 are impractical for TIP removal considerations.

The plots of the individual runs indicated that very little additional TIP removal was obtained in using Al:P molar ratios above 2.0. For example, in Figures 17 and 18 the effluent TIP values were approximately the same, 0.3 mg/l TIP, although the Al:P molar ratio ranged from 1.05 to 3.87. (Figure 19). The effluent phosphorus quality ranged from 0.5 to 0.9 mg/l TIP above the 1.4 Al:P molar ratio (Figure 20).

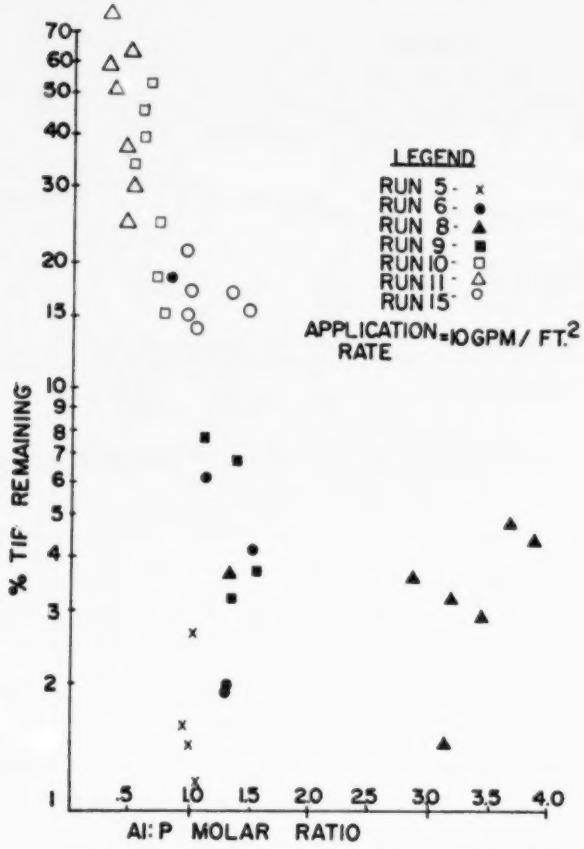

FIGURE 16. PERCENTAGE OF PHOSPHORUS REMAINING (ALL RUNS COMBINED SEWER OVERFLOW)

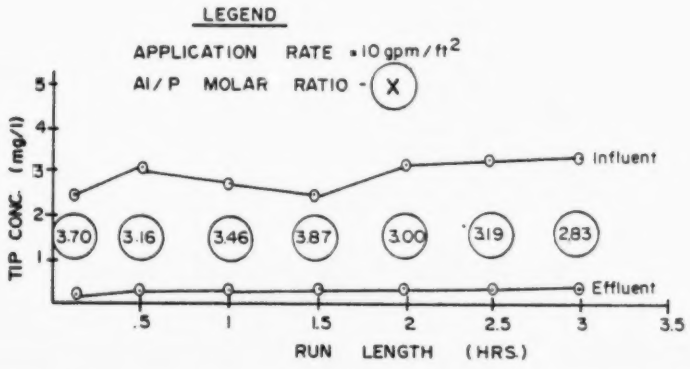

FIGURE 17. PHOSPHORUS REMOVAL (COMBINED SEWER OVERFLOW)

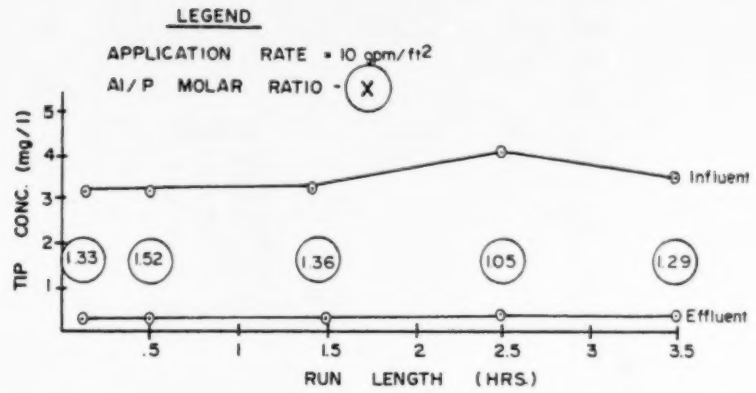

FIGURE 18. PHOSPHORUS REMOVAL RUN # 9 (COMBINED SEWER OVERFLOW)

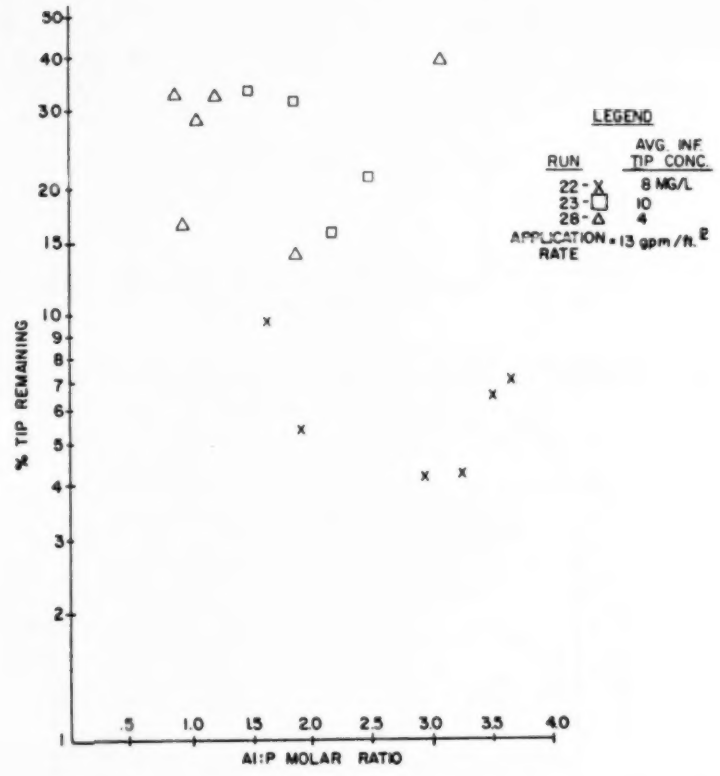

FIGURE 19. PERCENTAGE OF PHOSPHORUS REMAINING (COMBINED SEWER OVERFLOWS)

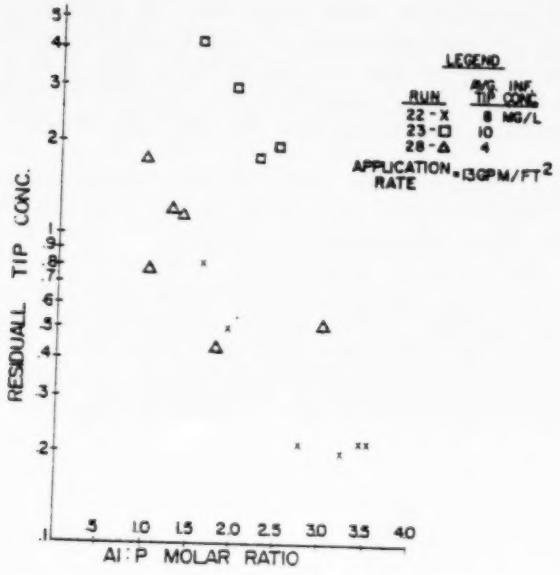

FIGURE 20 RESIDUAL PHOSPHORUS CONCENTRATION (COMBINED SEWER OVERFLOW)

Minton and Carlson (12) showed the results of various researchers in a plot of residual phosphorus versus Al:P molar ratio. The plot indicated that the results were quite variable in the Al:P molar ratio range of 1.0-4.0. In their presentation, an Al:P molar ratio of 2.0 or more generally produced an effluent with a filtrate phosphorus residual of 0.5 mg/l or less.

In Figure 20, the data indicated that the residual TIP concentration was unaffected by the Al:P molar ratio, when that ratio was above 1.4. With the exception of three data points all the residual TIP concentrations ranged from 0.3 - 0.9 mg/l. This suggested that there would be no advantage to using very high Al:P molar ratios to minimize the effluent phosphorus concentration.

Low effluent phosphorus concentrations encountered at minimum flux at the termination of runs reinforced the view that the in-line static mixers provided sufficient velocity gradients for the effective mixing of chemicals.

Ammonia Nitrogen Removal

The results of the ammonia evaluations are shown in Figures 22 through 25. These figures show the influent and effluent NH_3-N concentrations versus throughput volume for the various testing series.

The effective ammonia exchange capacity was based on a dry weight bulk density that was determined to be 762 kg/m^3 (47 lbs/cf) Hector clinoptilolite.

A preliminary set of 6 runs (runs No. 64 through 67 and No. 1 and No. 2) was made prior to the addition of any synthetic ammonia solution to the wastewater. The depth of the clinoptilolite was 1.52 m (5 feet) having a bed volume of 0.44 m^3 (15.7 $feet^3$). The initial application rate was 407 $l/min/m^2$ (10 gpm/ft^2) equivalent to an application rate of 15.9 BV/hr.

In these preliminary runs the influent NH_3-N concentration ranged from 0.20 - 0.97 mg/l NH_3-N, with a weighted average influent concentration of 0.37 mg/l NH_3-N as presented in Figure 21. The effluent NH_3-N concentration, initially below detectable levels, gradually rose to 0.2 mg/l NH_3-N. The total NH_3-N removed by the clinoptilolite was only 37.2 grams, which represented a partial EEC of 0.04 meq NH_3-N/g clinoptilolite. Although the column was not operated to exhaustion, this preliminary test did show that even at very low influent NH_3-N concentrations the clinoptilolite was very effective in the removal of NH_3-N.

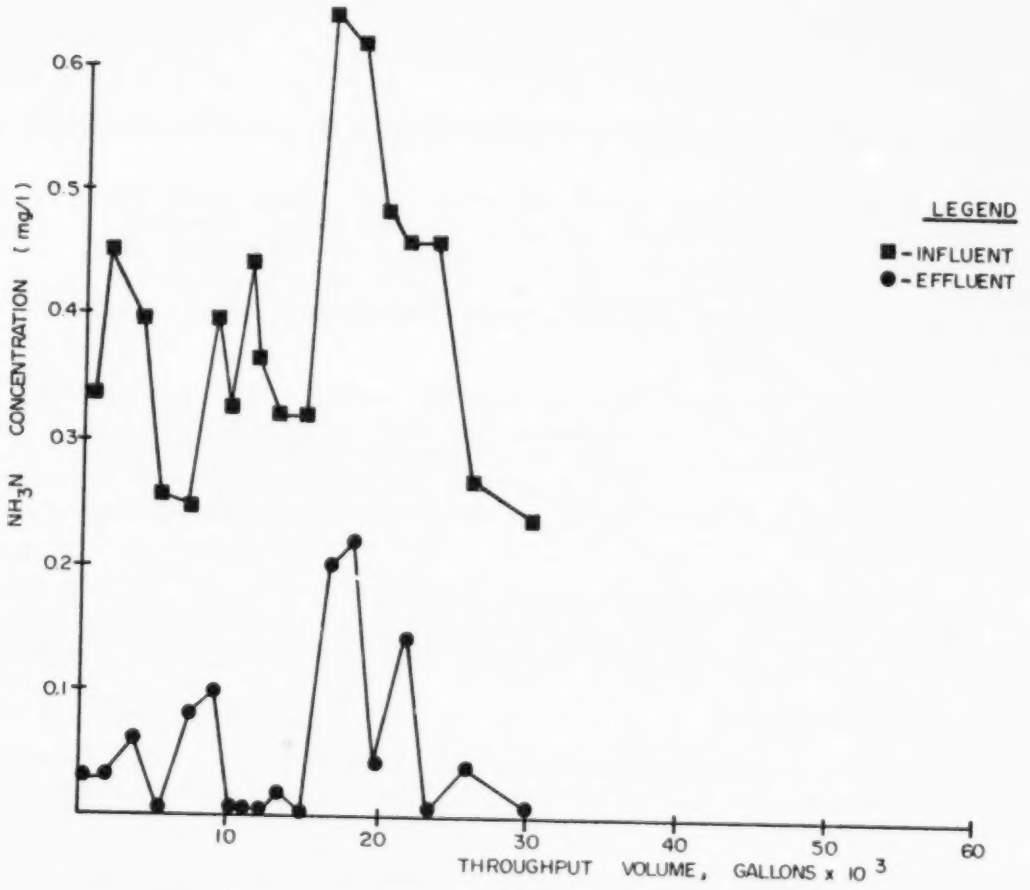

FIGURE 21. AMMONIA-NITROGEN REMOVAL BREAKTHROUGH CURVE PRELIMINARY TESTING CURVE

First Testing Series--

The first testing series consisted of 13 individual runs applied to a single column of clinoptilolite. The depth of the clinoptilolite was 1.52 m (5 feet) giving a bed volume 0.44 m^3 (15.7 ft^3) and the initial column application rate was 407 l/min/m^2(10 gal/ft^2/min) equivalent to an application rate of 15.9 BV/hr.

As can be seen in Figure 22, the influent NH_3-N concentration was quite variable ranging from 2.0 mg/l to 17.8 mg/l with a weighted average of 7.5 mg/l NH_3-N. The wide variation in influent NH_3-N concentration reflected operational problems experienced with the synthetic nutrients feed pump.

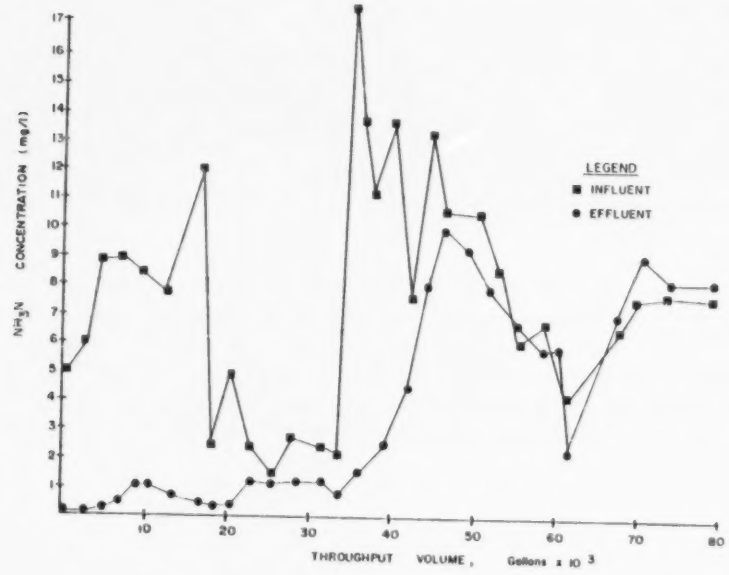

FIGURE 22. AMMONIA-NITROGEN REMOVAL BREAKTHROUGH CURVE FIRST TESTING SERIES

A total of 159 m^3 (42,000 gallons) had been applied to the column when the effective ammonia exchange capacity was reached, which is that point in the course of the run when the effluent and influent NH_3-N concentrations were nearly equal. At that point both influent and effluent concentrations were nearly 10 mg/l NH_3-N (the weighted average influent NH_3-N concentration was slightly lower).

It would appear that the breakthrough to 1 mg/l NH_3-N had occurred at 38 m^3 (10,000 gallons), 85 BV; however, the effluent ammonia concentrations then decreased to as low as 0.5 mg/l NH_3-N at 76 m^3 (20,000 gallons), BV. From the 76 m^3 (20,000 gallon) mark to the termination of the run at 185 m^3 (49,000 gallons) the effluent concentration gradually increased to

approximately 10 mg/l NH_3-N. Unfortunately, the influent NH_3-N concentration was too variable to yield a smooth breakthrough curve; however, the test series demonstrated the ability of clinoptilolite to effectively remove the NH_3-N. After 185 m^3 (49,000 gallons) had been applied, 964 g of NH_3-N had been subsequently removed. The effective ammonia exchange capacity of the clinoptilolite in this testing series was 2.88 mg NH_3-N/g clinoptilolite, which is equivalent to 0.20 meq NH_3-N/g clinoptilolite.

When the column of clinoptilolite reached its effective ammonia exchange capacity after the application 159 m^3 (42,000 gallons) of wastewater, an additional 113 m^3 (40,000 gallons) 340 BV were applied to the column. As can be seen in Figure 22, there was no additional observed NH_3-N removals, on the contrary there appeared to be some desorption occurring.

Second Testing Series--

The second testing series consisted of 10 separate runs using two columns of clinoptilolite in series. Both columns of clinoptilolite were kept on-line until the second column had reached its effective ammonia exchange capacity. The depth of clinoptilolite in both columns was the same, 1.22 m (4 feet) giving a media bed volume of 0.35 m^3(12.4 ft^3). The initial influx rate in this testing series was 529 l/min/m^2 (13 gpm/ft^2) equivalent to 25.5 BV/hr.

As can be seen in Figure 23, the influent concentration was somewhat variable, although less than observed in the previous testing series. The influent spiked NH_3-N concentration applied to the first clinoptilolite column ranged from 6-17 mg/l, with a weighted average influent of 9.4 mg/l NH_3-N. The influent ammonia concentration applied to the second clinoptilolite column was the effluent concentration from the first contactor.

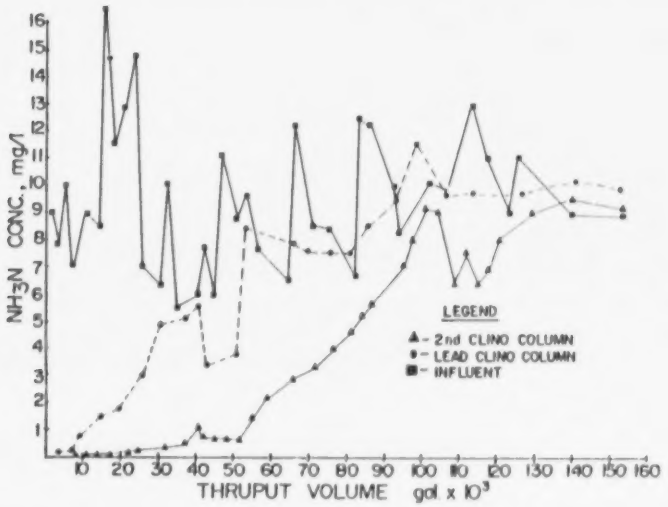

FIGURE 23. AMMONIA-NITROGEN REMOVAL BREAKTHROUGH CURVE SECOND TESTING SERIES

A total of 530 m^3 (140,000 gallons) of wastewater was applied to the contactors before the effluent from the second clinoptilolite column was approximately equal to the weighted average influent concentration applied to the same column.

The benefit of using two columns in series became readily apparent in this testing series. In the first column breakthrough to 1 mg/l NH_3-N occurred after 42 m^3 (11,000 gallons), 117 BV had been applied to the column. However, with the use of the second column, the volume at breakthrough from the entire two column series was increased almost fivefold to approximately 200 m^3 (53,000 gallons) or 281 BV based on the combined volumes of both columns. While this increase was not unexpected, it illustrated the value of using columns of clinoptilolite in a series mode of application.

When breakthrough to 1 mg/l NH_3-N occurred in the first column, 42 m^3 (11,000 gallons) of wastewater or approximately 355 g (0.8 lb) of NH_3-N had been removed by the clinoptilolite. When breakthrough at 1 mg/l NH_3-N occurred in the second column, 200 m^3 (53,000 gallons), an additional 895 g (1.9 lb) of NH_3-N was removed by clinoptilolite in the first column, bringing the total amount of NH_3-N removed to 1250 g. This quantity of ammonia removed was equivalent to 0.33 meq NH_3-N/g clinoptilolite.

In operating the two columns until the effluent NH_3-N concentration from the second column was approximately equal to the influent weighted average NH_3-N concentration, the first column had 5927 g (13.1 lb) NH_3-N applied and (3.0 lb) NH_3-N removed. Therefore, under the conditions of this testing series, the effective ammonia exchange capacity of the clinoptilolite in the first column was 5.12 mg NH_3-N/g clinoptilolite, equivalent to 0.37 meq NH_3-N/g clinoptilolite, considerably exceeding that realized with the second column.

In comparing the quantity of NH_3-N removed by the first clinoptilolite column upon breakthrough to 1 mg/l NH_3-N to that removed by the second column (0.09 vs 0.33 meq NH_3-N/g clinoptilolite), it is readily apparent that a much larger portion of the clinoptilolite in the first column had reached its effective exchange capacity before the breakthrough was experienced in the second column.

This follows the pattern of any typical ion exchange system where the breakthrough concentration front moves down through the media until the effluent concentration then begins to approach the influent concentration. It was apparent that the first column was at nearly 91% of its effective ammonia exchange capacity when breakthrough occurred in the second clinoptilolite column. During the entire testing series the second clinoptilolite column had 4554 g (10.0 lb) NH_3-N applied of which 1430 g (3.1 lb) NH_3-N was removed. The effective ammonia exchange capacity of the clinoptilolite in the second column was 5.33 mg NH_3-N/g clinoptilolite, equivalent to 0.38 meq NH_3-N/g clinoptilolite. Within the experimental error the two columns had the same effective adsorption capacity.

Even though the NH_3-N concentration being applied the second clinoptilolite column was quite low (less than 1 mg/l) during the first third of

the testing series, as the influent NH_3-N concentration to that column increased (simulating moving to the lead column in the series), the driving gradient became large enough to achieve the effective exchange capacity.

After the first clinoptilolite column had been exhausted, it still had the capacity to reduce any peak influent NH_3-N concentrations that were higher than the weighted average influent of 9.4 mg/l. However, any time the influent NH_3-N concentration dropped much below 7-8 mg/l NH_3-N, desorption was experienced.

The effluent from the second column exhibited a lower rate of NH_3-N increase breakthrough than the first clinoptilolite column. The reason for the rate of increase was due to low initial influent concentrations to the second column which gradually increased to being uniform throughout the testing series.

Third and Fourth Testing Series--

The third and fourth testing series were evaluated together to determine the effect of contact time on NH_3-N removal capacity of the clinoptilolite. A comparison of the results is shown in Figure 24. Both series were operated in the downflow mode through single columns of clinoptilolite. Creek water, spiked with synthetic nutrients, was used directly because the low suspended solids concentration in the creek water eliminated possible column plugging. The only parameter which was varied during the two testing series was the application rate. The depth of the clinoptilolite bed in both testing series was 1.37 m (4.5 ft), yielding a bed volume of 0.39 m^3 (14.1 ft^3).

The third and fourth testing series were operated at application rates of 326 l/min/m^2 (8 gpm/ft^2), and 65 l/min/m^2 (1.6 gpm/ft^2), respectively. Although specific influent NH_3-N concentrations to both columns varied somewhat, the weighted average influent NH_3-N concentrations in both testing series was quite similar. Although the effluent NH_3-N concentration was not quite equal to the weighted average influent NH_3-N concentration at the time of termination, Koon (6) has shown that the clinoptilolite breakthrough curve approached equilibrium (effective exchange capacity) relatively slowly, thus the effective adsorption capacity of the clinoptilolite at the end of this test series was likely to be very close to the capacity that would have been obtained had the effluent reached the 16 mg/l NH_3-N concentration. A total of 4140 (9.1 lb) g NH_3-N had been applied to the clinoptilolite column with 2712 g (5.9 lb) NH_3-N being removed. The effective ammonia removal capacity was 9.0 mg NH_3-N/g clinoptilolite, equivalent to 0.62 meq NH_3-N/g clinoptilolite. At the conclusion of the test, the clinoptilolite was not fully exhausted. After 246 m^3 (65,000 gal) had passed through the clinoptilolite, 3324 g (7.3 lb) NH_3-N, of 3994 g (8.8 lb) NH_3-N applied, were removed by the clinoptilolite resulting in an EEC of 0.79 meq NH_3-N/g clinoptilolite. Since the effluent NH_3-N concentration at the end of this test was significantly lower than the influent weighted average of 16 mg/l NH_3-N, a significant quantity of ammonia would probably continue to be removed by the clinoptilolite bed (assuming that breakthrough to 1 mg/l NH_3-N represented approximately 65% of the removal capacity), as in the third test. The effective ammonia removal

capacity in the fourth test could then be expected to approach 13.3 mg NH_3-N/g clinoptilolite, equivalent to 0.95 meq NH_3-N/g clinoptilolite.

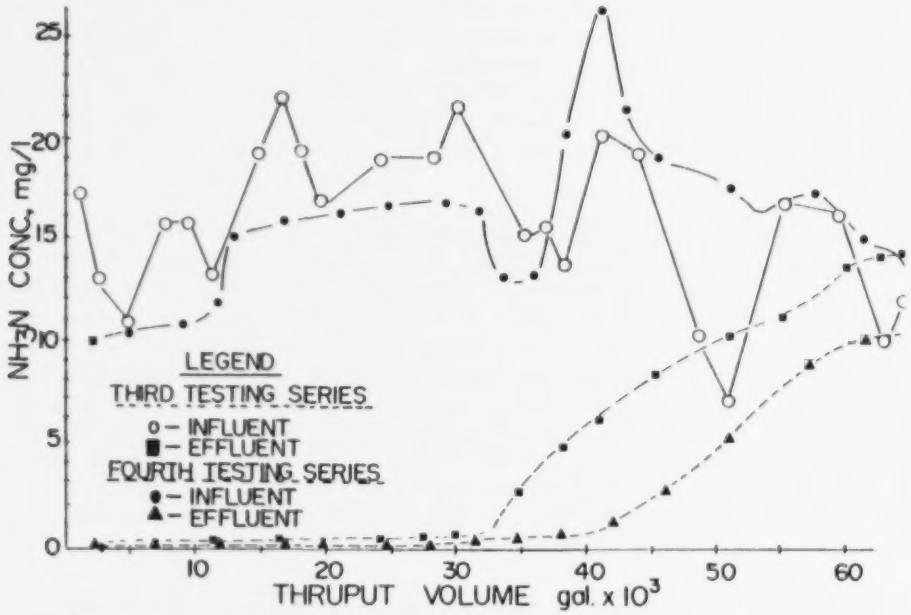

FIGURE 24. AMMONIA-NITROGEN REMOVAL BREAKTHROUGH CURVE THIRD AND FOURTH TESTING SERIES

Comparing the effect of the two application rates on the ammonia removal capacity, particularly at breakthrough to 1 mg/l NH_3-N, it was evident that at the lower flow rate (increased residence time) an additional 38 m^3 (10,000 gal) was passed through the column before breakthrough occurred. Also, it would appear that additional NH_3-N can be adsorbed on the clinoptilolite at the lower flow rate. The results show that almost a 50% increase in the adsorption capacity can be realized with a 80% decrease in the application rate. The tests indicated that the residence time of the wastewater in the clinoptilolite bed does affect adsorption; however, the relative economy associated with the higher application rate may offset the increase in the adsorption capacity.

The data indicated that the clinoptilolite was very effective in removing the ammonia from the applied wastewater. The influent NH_3-N concentrations, ranging from 0.2 mg/l to 16 mg/l, during the range of testing series, were reduced below detectable levels in the initial period of contact. The degradation of NH_3-N effluent quality followed the exhaustion of the available active sites.

Removal of Other Wastewater Constituents

It is common during filtration that in addition to suspended solids, other constituents in the applied wastewater are also removed.

If chemical addition is employed, the suspended and colloidal solids can be swept out of solution by a "sweep floc" and the soluble constituents can be removed from solution by coprecipitation. During one particular run, in addition to the regularly scheduled analyses of the influent and effluent samples, several other parameters were monitored. The parameters included heavy metals: cadmium, chromium, copper, iron, lead, and zinc, as well as, TOC and fecal coliform bacteria. The results of the analyses are listed in Table 11. The influent concentration of lead and cadmium were below the analytical detection limits, and as a result they are not presented in Table 11.

The presented data illustrates that there is a reduction in heavy metals concentrations, even though they are initially present in rather low quantities. The best removal was obtained with iron, which consistently showed removals of greater than 99%. Removal percentages for other heavy metals ranged from 0% to > 99%, and averaged usually better than 40% removal. The lack of correlation between the SS and the heavy metal reductions eliminated coprecipitation as a possible removal mechanism for heavy metals. Similarly, the TOC reductions, which ranged from 28% to 75%, could not be correlated to SS reductions. Reductions in the fecal coliform concentrations in the combined sewer overflows were as high as 90%. Influent fecal coliform levels averaged in the 500,000 colonies/100 ml range and were reduced in many cases to below 10,000 colonies/100 ml.

The entrainment of bacteria in the alum floc and its subsequent removal in the filter should greatly reduce the disinfectant requirements of the CSO, should further bacterial reductions be necessary. This would result in lower disinfectent operating costs and a reduction in the possibility of high oxidant residual levels being discharged to the aquatic environment.

DRY WEATHER CREEK FLOW TREATMENT

During periods of dry weather, creek water was applied to the nutrient removal system. The original intent of this application was to determine if the wet weather facilities could be utilized during dry weather periods to reduce the loading of suspended solids, phosphorus and ammonia nitrogen to Onondaga Lake. The concentrations of suspended solids, phosphorus, and ammonia-nitrogen measured in the creek water during the study were found to be quite low: 20-75 mg/l SS, 0.1-0.4 mg/l TIP, and 0.2-0.7 mg/l NH_3-N, respectively. Only the SS concentration was found to increase to rather high levels, a maximum of 100-120 mg/l SS following an intense rainfall. The TIP and NH_3-N remained within the previously staged range. Since the background levels of TIP and NH_3-N in the creek water were so low, dibasic ammonium phosphate was added to increase the levels to approximately 10 mg/l TIP and 10 mg/l NH_3-N for the demonstration scale testing.

Of the 13 runs made using creek water only two resulted in suspended solids removal efficiencies consistently greater than 25%. Poor SS removal efficiencies were probably the result of low SS concentrations in the applied creek water and the low alum dosages, 50, and 100 mg/l

TABLE 11. ADDITIONAL ANALYSES FOR A COMBINED SEWER OVERFLOW

Flux: 407 l/min/m^2 (10 gpm/ft^2) Media: #3 Anthrafilt Media Depth: 7 ft
Alum Dosage: 215 mg/l Polymer Dosage: 1.0 mg/l

Sample	Run Time hr.	Flow Rate lpm (gpm)	pH	Cr mg/l	Cu mg/l	Fe mg/l	Zn mg/l	TOC mg/l	TSS mg/l	FColi. col/100 ml
Influent	0.0	117.3(31)	8.1	0.05	0.04	0.37	0.06	32.	42.	665,000
Effluent	0.1		6.9	<0.01	0.02	<0.01	0.03	23.	3.	305,000
Influent	0.7									660,000
Effluent	0.8									7,400
Influent	1.0	107.9(28.5)	7.9	0.03	0.04	0.60	0.07	50.	50.	545,000
Effluent	1.1		6.9	<0.01	0.02	<0.01	0.05	24.	10.	5,850
Influent	1.3									530,000
Effluent	1.4									2,553
Influent	1.7									395,000
Effluent	1.8									3,750
Influent	2.0	98.4(26)	7.7	0.03	0.02	0.57	0.06	55.	116.	535,000
Effluent	2.1		6.8	<0.01	0.02	<0.01	0.03	28.	20.	5,850
Influent	2.7	81.4(21.5)	7.6	0.03	0.02	0.57	0.07	60	50.	242,700
Effluent	2.8		7.0	<0.01	0.00	<0.01	0.04	14	51.	17,850

alum, used as a coagulant. Due to the dispersed and stabilized state of the total suspended solids characteristic of the creek water large alum dosages would be required to effect a high removal of SS. However, this would not be a cost-effective practice. In this case, suspended solids removal could possibly be improved by using a smaller media size in the filters.

In both creek water runs considered to have good SS removals, the chemical dosages were 100 mg/l alum and 4 mg/l polymer. It is believed that, in these two cases, the very high polymer dosage was instrumental in achieving the superior SS removals. Other creek water runs involved the application of equal and even greater alum dosages but lower polymer dosages and achieved inferior SS removals. Another factor which could help explain the improved SS removals realized in the two runs is that, of all the creek water applications these two had the highest influent SS loading. In one run, the influent SS ranged from 55-115 mg/l, while the effluent ranged from 18-30 mg/l for a SS removal efficiency of 25-88%. The run was 2.5 hours in length, and with the flow rate decreasing from 1.96 l/sec (10 gpm/ft^2) to 1.33 l/sec (6.8 gpm/ft^2) during the run, the total throughput volume was 14800 l (3900 gallons). Termination of the run after 2.5 hours was necessary because of chemical feed system failure.

For the second run influent SS ranged from 30-70 mg/l, while the effluent ranged from 2-6 mg/l SS, representing a removal efficiency of 91-97%. After a 3.5 hour run length, the flow rate had decreased from 1.96 l/sec (10 gpm/ft^2) to 1.14 l/sec (5.7 gpm/ft^2) with a total volume of (5200 gallons) subsequently processed through the filter.

For the remaining creek water runs SS removal efficiencies were erratic and inconsistent. Influent suspended solids concentration in these eight runs were lower, ranging from 5-50 mg/l with applied alum and polymer dosages ranging from 50-200 mg/l and 1-2 mg/l, respectively.

Since treatment of creek water from the standpoint of SS removals appeared to be impractical, increased alum and polymer dosages were not applied in an attempt to increase the removal efficiency of the SS.

Phosphorus removals experienced on the creek water were more than acceptable and were similar to the removal efficiencies experienced in the combined sewer overflow. Analysis of the data provided no trends unique to the creek water runs. The reason may be partially due to the synthetic nutrient solution that was added to both the combined sewer overflows and the creek water. The data indicated that Al:P molar ratios above 1.4 generally resulted in 80-95% TIP removal, equivalent to a residual TIP concentration of 0.1 to 0.8 mg/l TIP. As with the overflow occurrences, an Al:P molar ratio below 1.0 yielded poor TIP removals.

The ammonia-nitrogen removal efficiencies realized in the creek water runs were essentially the same as those experienced in the combined sewer overflow runs. There was no indication that the ammonia-nitrogen removal mechanisms from creek water differed in any from the ammonia removal experienced with the overflows.

SECTION 7

REGENERATION EVALUATIONS

CLINOPTILOLITE REGENERATION

The high cost of clinoptilolite makes the regeneration of clinoptilolite a necessary alternative to the disposal of exhausted material. Current regeneration technology involves the use of a caustic or caustic-brine regenerant. However, for this process to be economically practical, the regenerant must be stripped of ammonia and recycled. The removal of ammonia from the regenerant is effected at an elevated pH in a stripping tower. Because the operation of a stripping tower is limited to climatic conditions where no freezing occurs during winter, its application is, therefore, not very practical in the northeastern and midwestern U.S. An alternate method of clinoptilolite regeneration was investigated.

Thermal Regeneration

Thermal regeneration evaluations were conducted on Hector clinoptilolite samples at temperatures of 300°C, 400°C, 500°C, and 600°C in a laboratory rotary tube furnace (Linberg). Figure 25 shows a schematic diagram of the furnace apparatus.

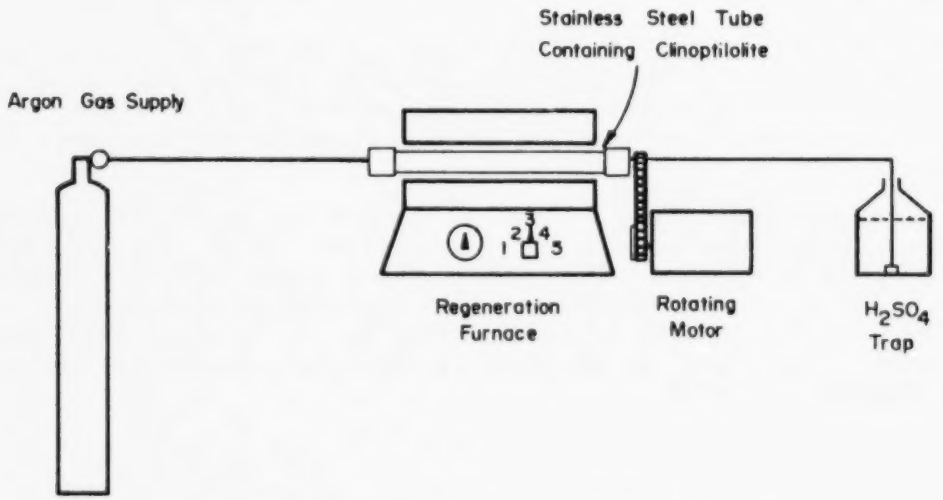

FIGURE 25. ZEOLITE REGENERATION SYSTEM

The spent clinoptilolite samples were placed in a stainless steel tube which could be inserted into the furnace. The furnace apparatus was configured so a carrier gas could be passed through the stainless steel tube thereby conveying the off-gases through a trap containing 500 ml of 120 g/l H_2SO_4.

The exchange capacity of the clinoptilolite was determined by passing a concentrated NH_3-N solution (3,000 mg/l NH_3-N) through a column containing 400 g (0.9 lb) of clinoptilolite until the effluent NH_3-N concentration was equal to the influent NH_3-N concentration. Following exhaustion, two liters of demineralized water were passed through the column to remove any residual NH_3-N not actually adsorbed on the clinoptilolite. Samples of both the influent and effluent were taken and analyzed for NH_3-N to determine the exchange capacity per gram of clinoptilolite.

A 15 g sample of the dried, spent clinoptilolite was placed in the stainless steel tube and inserted into the furnace, previously heated to the desired temperature. Air was blown through the tube at a constant rate of one l/min while rotating the tube at 10 rpm. Samples from the ammonia trap were taken initially and every 15 minutes thereafter for two hours. From the data, the amount of NH_3-N desorbed could be determined.

The results of the thermal regeneration using compressed air as the carrier gas were non-reproducible. The variation in the results were most likely caused by the oxidation of ammonia within the temperature range of 500°C-600°C in an oxygen atmosphere. A source of the variation may stem from the conversion of an undeterminable quantity of desorbed ammonia to N_2 before being trapped in the 120 g/l H_2SO_4 solution. Unfortunately, the analytical control was not designed to delineate this mechanism and it was therefore impossible to determine the exact quantity of ammonia oxidized at the higher temperatures.

The highest percentage of recovered adsorbed ammonia was only 53% at 500°C. The results also show that only 35% of the adsorbed ammonia was recovered at 600°C, suggesting that the difference may be due to the oxidation of a larger fraction of the ammonia at the further elevated temperatures.

Since significant variations were observed in the ammonia recovery in the thermal regeneration tests using compressed air, an inert gas (argon) was used as the carrier gas. The results obtained using argon are presented in Figure 26. As shown previously, thermal regeneration was much more effective at 500°C and 600°C than 300°C or 400°C. At 300°C and 400°C only 25% and 65%, respectively, of the NH_3-N adsorbed onto the clinoptilolite was desorbed and recovered.

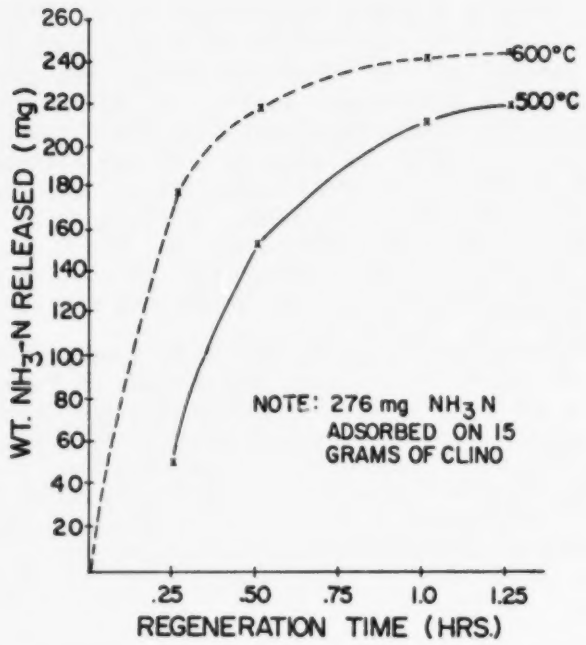

FIGURE 26. AMMONIA RECOVERY FROM THERMAL REGENERATION OF CLINO

The NH_3-N uptake capacities of the regenerated clinoptilolite samples are presented in Table 12. With an influent ammonia concentration of approximately 13 mg/l, the effective exchange capacity of the virgin clinoptilolite was 0.228 meq NH_3-N/g while the clinoptilolite, thermally regenerated at 500°C and 600 C had effective exchange capacities of 0.244 and 0.203 meq NH_3-N/g clinoptilolite, respectively.

TABLE 12. EFFECTIVE ADSORPTION CAPACITY OF THERMALLY REGENERATED CLINO

Sample	mg NH_3-N Adsorbed	meq NH_3-N/g Clinoptilolite
Virgin Clinoptilolite Hector	6.38	0.228
300°C Regenerated Clinoptilolite	-24.63	-
400°C Regenerated Clinoptilolite	- 3.37	-
500°C Regenerated Clinoptilolite	6.86	0.245
600°C Regenerated Clinoptilolite	5.71	0.204

It should be noted, that the sample regenerated at the 500°C had a slightly higher EEC than the virgin clinoptilolite which was used as a control. This may be due to the removal of some impurities within the clinoptilolite, such as carbonates or volatile adsorbates, during the thermal treatment. The slightly lower EEC of the sample regenerated at 600°C could have been caused by an alteration in the crystalline structure of the clinoptilolite. This is supported by the reports of several investigators who reported that the clinoptilolite crystalline structure begins to collapse at 750°C (13). Samples regenerated at 200°C and 400°C showed no NH_3-N uptake when contacted with the influent NH_3-N solution. Conversely, these samples released NH_3-N into the contacting solution, indicating incomplete regeneration.

As part of this study, X-ray diffraction analyses of Hector clinoptilolite samples were performed following the methods of Mumpton (13) and Rooney and Kerr (14). The analysis of virgin, exhausted, and thermally regenerated (500°C) Hector clinoptilolite samples indicated little or no structural degradation of the thermally regenerated clinoptilolite. Several x-ray power pattern spectra are shown in Figure 27. From Figure 27 it appears that the effect of thermal regeneration results in the broadening and elimination of a number of diffraction spectral lines indicating the loss of adsorbed constituents, occluded water, and possibly some minimal structural alteration.

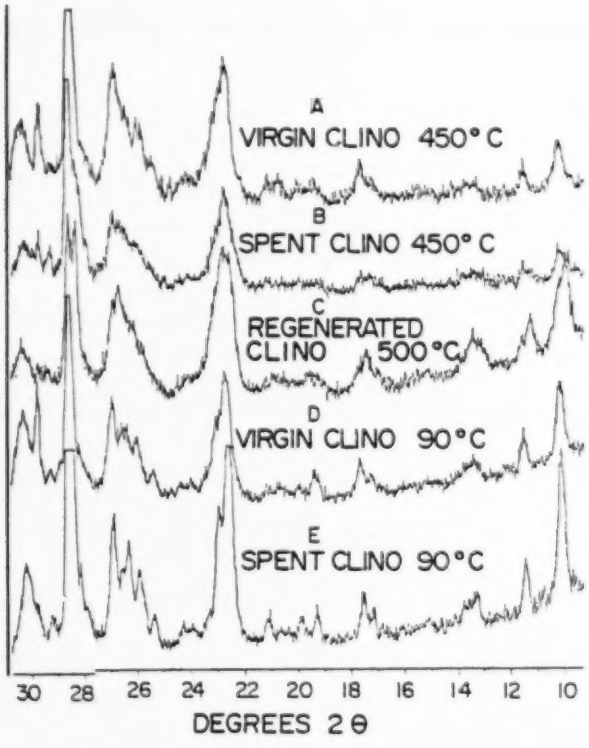

FIGURE 27. X-RAY DIFFRACTION PATTERNS OF CLINOPTILOLITE

ALUM REGENERATION

There are two methods of alum recovery, acid treatment and alkaline treatment of the alum sludge followed by a solids separation process. Excellent recoveries of alum were reported by Slechta and Culp (15) when they lowered the pH of the alum sludge to 2.5. Unfortunately, phosphorus was also recovered, and the authors concluded that alum recovery was feasible only when phosphorus removal was not required.

The alkaline treatment consisted of elevating the pH of the waste alum sludge with either lime or sodium hydroxide followed by treatment with calcium chloride. Lea, et al (16) reported that sodium hydroxide and calcium chloride treatment to pH 11.9 resulted in a high percentage of recovered aluminum (93%); however, the coagulation properties of the reclaimed aluminum were inferior to fresh alum when used in dosages resulting in an Al:P molar ratio of 2. Coagulant reclamation via lime treatment of the alum sludge to pH 10.2 was reported by Slechta and Culp (15) to be ineffective as only 20% of the aluminum was recovered.

The approach taken in this study was to thermally treat the spent alum sludge and then dissolve it in sulfuric acid. It was hypothesized that at elevated temperatures the phosphorus in the alum floc would possibly convert to phosphorus pentoxide, P_2O_5 (phosphorus anhydride) which volatilizes at about 250°C. The phosphorus pentoxide would then be trapped utilizing a caustic scrubber, where it would form sodium phosphate $Na_3(PO_4)_2$. At temperatures approaching 900°C the hydrated aluminum hydroxide floc may dehydrate and convert to a partially hydrated aluminum oxide, $Al_2O_3 \cdot XH_2O$.

Sufficient dibasic potassium phosphate was dissolved in distilled water to yield a phosphate concentration of 1.2%. To this solution, alum was added until a floc was formed. The floc was allowed to settle and the clear supernatant was decanted and analyzed for phosphate. The resultant sludge was centrifuged and dried at 103°C for an hour. A portion of the dried sludge was placed in a furnace, preheated to 925°C, for a period of 20 minutes. The sludge samples were cooled, weighed, and digested in 10 ml of hot sulfuric acid, H_2SO_4, and 2 ml of concentrated nitric acid, NHO_3. After 2.5 hours of digestion, the samples were cooled, and the liquid level was brought up to 50 ml in preparation for phosphorus analyses.

Analyses of the resultant data indicated that thermal treatment of the waste alum sludge at 925°C did not reduce the amount of phosphate recovered. On the contrary, more phosphate was recovered from the alum sludge treated at 925°C for 20 minutes, than from the alum sludge treated at 105°C. Almost twice as much phosphate (PO_4) was recovered from the 925°C treatment than from the 105°C treatment. The actual percentages of PO_4 recovered were 28.3% and 14.4% for the alum sludge treated at 925°C and 105°C, respectively.

Several other tests were conducted for the purpose of evaluating the effect of several temperatures on PO_4 recovery. The temperatures evaluated

were 105°C, 250°C, 500°C, 800°C, and 1000°C. These tests differed from the initial evaluations in that domestic sewage was used rather than a phosphate solution. The samples were dosed with 150 mg/l alum which resulted in an aluminum to phosphorus molar ratio of 5.2. The procedure used was essentially the same as described above, with the exception that the sludge was filtered rather than centrifuged. The digestion step involved the use of 5 ml of concentrated H_2SO_4 and 15 ml of water.

The data from this series of tests confirmed the conclusions drawn from the earlier tests that thermal treatment of alum sludge does not reduce the amount of phosphate recovered during its acidification. An interesting observation noted during these tests was that essentially all of the phosphate was recovered from the alum sludge regardless of the temperature of treatment. Because of the high levels of phosphate recovered upon acidification of the waste alum sludge, the percent recovery of aluminum was not determined and its effectiveness as a coagulant was not further evaluated.

SECTION 8

PROJECTED SYSTEM COSTS

GENERAL

In developing a cost estimate for a high rate combined sewer overflow treatment system for nutrient removal, a number of assumptions must be made in designing such a system to treat a "typical' waste. It must be noted that the costs developed, herein, are only estimates, and should be considered only as a guideline when contemplating the installation of a similar system. Although major equipment items may have similar costs, installation costs and particularly conveyance costs are site dependent. Therefore, depending on the exact location the actual costs may be either higher or lower.

CAPITAL COSTS

Capital costs are based on a system designed to handle a combined sewer overflow occurrence with the following characteristics:

Number of overflows per year	100
Design capacity	10 mgd
Duration treatment facility operation	4 hours
Average treatment rate	5 mgd
Duration of overflow	8 hours
Influent quality:	
Average SS conc.	200 mg/l
Average NH_3-N conc.	15 mg/l
Average TIP conc.	10 mg/l

The operating characteristics of the proposed system were developed from the demonstration scale study and are listed in Table 13.

TABLE 13. OPERATING CHARACTERISTICS FOR A 10 MGD COMBINED SEWER OVERFLOW NUTRIENT REMOVAL FACILITY

Horizontal drum screen size	- 0.305m x 0.035m (10 ft. ø by 10 ft.)
Horizontal drum screen application rate	- 1 m^3/min/5 gm (25 gpm/ft^2)
Filtration application rate (variable flow)	- 456 l/min/m^2 (11.2 gpm/ft^2)
Filter solids capacity	- 5.6 kg SS/m^3 (0.35 lb SS/ft^3- includes alum SS)
Design for peak flow rate	- 37.85 x 10^3 m^3/day (10 mgd)
Filter area (total)	- 65 m^2 (700 ft^2)
Number of filters (pressure)	- 4
Filter media depth	- 2.5 m (8 ft.)
Filter dimensions	- 15' dia. x 18' depth
Filter run	- 6 hours
Alum dosage	- 215 mg/l
Polymer dosage	- 1 mg/l polymer
Ion exchange/adsorption application rate	- 15 BV/hr
Clinoptilolite ammonia capacity	- 0.42 meq NH_3-N/g to breakthrough to 1 mg/l NH_3(N)
Clinoptilolite bed volume BV	- 105 m^3 (3750 ft^3)
Clinoptilolite bed depth	- 1.83 m (6 ft)
No. of contactors	- 12
Clinoptilolite contactor dimensions	- diameter: 4.27 m (14 ft) height: 4.57 m (15 ft)
Cycle time	- breakthrough lead column:22 hrs. exhaustion lead column:35 hrs.
Thermal regeneration time	- 7 days

The complete system is parcelled into several subsystems which are listed below, and the capital costs are presented in Table 14.

1. High Rate Filtration Plant Pumping Station
2. Screening Subsystem
3. Chemical Addition Subsystem
4. Filtration
5. Ion Exchange
6. Regeneration Facilities
7. Miscellaneous

TABLE 14. CAPITAL COSTS FOR 10 MGD NUTRIENT REMOVAL SYSTEM FOR CSO

1. Pumping Station	
a) Excavation and Hauling	$ 41,000
b) Reinforced Concrete	40,000
c) Flow Meters	20,000
d) Pumps and Control	54,000
e) Liquid Level Recorder	5,000
f) Miscellaneous	10,000
	$170,000

2. Screening Facilities	
3.05 m x 3.05 m (10 ft x 10 ft) Rotating Horizontal Drum Screen	$ 83,000
3. Chemical Facilities	
a) Alum tanks, mixer and pump	120,000
b) Polyelectrolyte tanks, mixer and pump	65,000
c) In-line static mixers	18,000
	$203,000
4. Filter System	
a) Filter with valving, etc.	250,000
b) Media (initial charge)	10,000
	$ 260,000
5. Ion Exchange System	
a) Contactors	400,000
b) Media (initial charge)	280,000
	$680,000
6. Thermal Regeneration Facility	
a) Conveyance system	50,000
b) Dewatering system	60,000
c) Multiple hearth furnace	750,000
	$860,000
7. Miscellaneous	
a) Building	250,000
b) Heating & ventilation	30,000
c) Electrical	30,000
d) Excavation and landfill	15,000
e) Filter pumps	60,000
f) Instrumentation	50,000
g) Piping	50,000
h) Site preparation	20,000
	$505,000
Subtotal	$ 2,761,000
Installation (40% subtotal)	1,105,000
	$ 3,866,000
Construction Contingency (15%)	580,000
	$ 4,446,000
Engineering & Administration (12%)	534,000
TOTAL ESTIMATED PROJECT COST	$ 4,980,000

ANNUAL COSTS

Annual operating and maintenance costs are projected in Table 15 on the basis of treating 100 CSO occurrences per year. This system is expected to operate at a peak flow for four hours and at an average flow for 8 hours. Assuming each occurrence lasts 12 hours, this results in a total of 1200 hours of facility operation per year. Although the CSO facility may be automated to such a degree that requires minimum manpower involvement, the thermal regeneration unit for spent clinoptilolite would be in operation 24 hours/day and would require moderate supervision.

The annual costs of the Nutrient Removal Facility are listed in Table 15.

TABLE 15. ANNUAL COST OF NUTRIENT REMOVAL FACILITY

Operating	
a) Alum	$27,000
b) Polyelectrolyte	8,500
	$35,000
a) Anthrafilt	$ 1,000
b) Clinoptilolite	56,000
	$57,000
Labor	60,000
Maintenance	
a) Mechanical (3% of Mechanical Equip. Costs)	$39,000
b) Electrical (2% of Electrical Equip. Costs)	15,000
c) Structural (1% of Structural Equip. Costs)	5,000
	$59,000
Utilities	
a) Electricity ($.03/KWH)	$12,000
b) Fuel Oil ($.40/gal)	10,000
	$22,000
Amortization	
7.5% interest rate for 25 years	$447,000
TOTAL ANNUAL COSTS	$680,000

As indicated in the above table the most significant cost is the amortization of the facility, which in this case is equivalent to 70% of the total annual costs. An accurate treatment cost per 1000 gallons treated can not be established since the system would not be operating at full capacity for the entire overflow occurrence. Under the conditions mentioned previously in this section, the facility would operate at 33% capacity resulting in a cost of $3.00 per 1000 gallons of CSO treated. It should be emphasized that these costs are based on projections for a certain geographical area, Syracuse, N.Y. Actual treatment costs per unit volume would vary with location, since they are dependent on factors, such as the number of rain events, duration and intensity of the rain events, size of drainage area, and physical condition of the combined sewers. Also, some economy may be realized if a centrally located thermal regeneration facility were available that would regenerate clinoptilolite from several overflow treatment sites.

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GLOSSARY

BED VOLUME PER HOUR (BV/hr.): A term used to simultaneously define flux and contact time for a removal process relying on either a sorption or exchange mechanism. Bed volume refers to the amount of exchange material in the contactor.

CLINOPTILOLITE: A naturally occurring zeolite which has an ion exchange specificity for the ammonium cation. It is stable upon dehydration and readily readsorbs water. It is thermally stable up to 700^{o} C.

EFFECTIVE EXCHANGE CAPACITY (EEC): The amount of substrate removed by an exchange material under dynamic conditions, expressed as milliequivalents of substrate removed per gram of exchange material (meq/g). Principal factor influencing the EEC of an exchange material is the initial substrate concentration.

EXCHANGE CAPACITY: The amount of substrate removed per unit weight of exchange material under equilibrium conditions, expressed as milli-equivalents of substrate removed per gram of exchange material (meq/g).

FLUX or INFLUX: Rate of liquid flow expressed as unit volume/unit time.

ZEOLITE: Crystalline, hydrated aluminosilicates of alkaline earth materials (sodium, potassium, calcium, magnesium, barium and strontium), having a three dimensional silicate structure.